University of Alberta

Fencerows as Habitat for Birds and Small Mammals

by

Ainsley Katherine Sykes

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Master of Science

in

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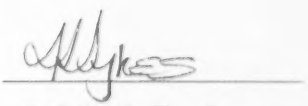

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This thesis is dedicated to Mark and Tara.....
To conquering all our "Scary Monsters"

ABSTRACT

Fragmentation of the landscape by agricultural practices results in a mosaic of fields interspersed with isolated woodlots and fencerows. During two summers (1993/1994), I examined factors affecting avian and small mammal abundance in 30 aspen-dominated fencerows in agricultural (20% wooded) and forested (48% wooded) landscapes. Landscape type was a primary predictor of small mammal abundance with *Peromyscus maniculatus* (deer mouse) and *Clethrionomys gapperi* (red-backed vole) abundances higher in the agricultural and forested landscapes, respectively. Isolation did not affect *P. maniculatus* abundance, however, *C. gapperi* abundances were higher in connected fencerows in the year with relatively higher densities. In contrast, only characteristics within fencerows affected avian species richness, abundance and territory density. Characteristics influencing the territory abundances of individual species are shown. Avian species found in fencerows were predominantly edge species and represented only a subset of the local species pool in the surrounding woodlots.

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"It is not the destination, it is the journey that makes you strong" (Anon).

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CHAPTER 1

Thesis Introduction

1.0 INTRODUCTION

Habitat fragmentation decreases the amount of natural habitat available to wildlife and reduces connectivity between the remaining fragments, resulting in the isolation of populations (Saunders *et al.* 1991). Declines in bird species, especially neotropical migrants, have been attributed to habitat loss and isolation (Lynch & Whigham 1984, Freemark & Merriam 1986, Robbins *et al.* 1989) and to the introduction of anthropogenic edge. Densities may be elevated at edges (Gates & Gysel 1978) but edges can function as ecological traps due to increased predation, competition and brood parasitism (Brittingham & Temple 1983, Noss 1983). Very small fragments may be entirely impacted by negative edge effects (Noss 1983). Negative edge effects probably affect small mammals in a similar way, however there is a paucity of studies investigating these effects (Mills 1995). In general, small mammal populations are positively affected by area of the fragment (e.g. Brown 1978, Bennett 1987 & 1990a, Mills 1995) and negatively affected by isolation (e.g. Gottfried 1979, Fahrig & Merriam 1985, La Polla & Barrett 1993).

Fragmentation of forest by agriculture results in a characteristic mosaic of crop fields interspersed with isolated woodlots and linear strips of vegetation along field edges in the form of shelterbelts, windbreaks, hedgerows and fencerows. Linear strips of trees and vegetation which either follow or originally followed a fence are called fencerows (Forman & Baudry 1984). The purpose of these linear strips is to provide protection to cattle and crops from wind, rain and snow, and to prevent soil loss.

The majority of fragmentation research focuses on woodlots (e.g. Lynch & Whigham 1984, Freemark & Merriam 1986, Robbins *et al.* 1989) and the role of linear strips as conduits between fragments (e.g. Bennett 1990b). Fencerows may, however, represent critical habitat in landscapes where few woodlots remain. In an agricultural landscape, the amount of wooded habitat available is important to the survival of many forest species that are unable to live in cultivated land (Middleton & Merriam 1983, Baudry 1988). Many species require large tracts of wooded habitat to persist and fencerows will be unable to support such species. Therefore, in an agriculturally fragmented landscape there may only be a subset of the species pool originally found in the region, and fencerows may only be able to support a portion of these remaining species. Due to the small size of fencerows, the populations will be small with increased probabilities of both extinction (Askins *et al.* 1987, Freemark & Merriam 1986) and negative edge effects occurring. Determination of the role of fencerows as habitat and the factors affecting their use by different species is important to assess the contribution of fencerows in the survival of species in agricultural landscapes.

At the turn of the century the area surrounding the Athabasca Landing in north central Alberta was virtually uninhabited (Stone 1969). The first rural settlers arrived in 1904 (Stone 1969) and since then the boreal mixed-wood forest has gradually been fragmented by agricultural settlers to the point today where approximately 55% is under cultivation. Now in this area of north central Alberta, the majority of the woodlots are privately owned and in the year prior to my study, a number of woodlots were logged by the pulp and paper industry in the region. Many of the remaining woodlots are used for

cattle grazing which alters vegetation through browsing and trampling and via indirect effects of compacting soil, injuring roots and affecting water retention (Weatherhill & Keith 1969). As a result of logging and browsing there are few undisturbed woodlots, thus reducing the habitat available to wildlife. Due to the paucity of woodlots, fencerows may be of considerable importance to birds and small mammals in this area.

The aim of this research was to determine which birds and small mammals inhabit fencerows and the features of the landscape and the fencerow which influence species richness and abundance. No research has been conducted on wildlife in fencerows in western Canada; the majority of research into linear strips has been conducted in eastern Canada (e.g. Henderson *et al.* 1985, Merriam & Lanoue 1990, Wegner & Merriam 1990), Europe (e.g. Pollard & Relton 1970, Burel & Baudry 1990, Zhang & Usher 1991) and the United States (e.g. Emmerich & Vohs 1982, Yahner 1983).

In chapter two, I examine the effect of landscape and fencerow characteristics on small mammal abundance in fencerows and in chapter three I examine the fencerow and landscape characteristics influencing avian species richness and abundance in fencerows. The characteristics under investigation include fencerow width and area, isolation, adjacent land use, vegetation, and landscape type and context.

1.1 LITERATURE CITED

Askins, R.A., M.J. Philbrick, and D.S. Sugeno. 1987. Relationship between the regional abundance of forest and the composition of forest bird communities. Biological Conservation **39**: 129-152.

Baudry, J. 1988. Hedgerows and hedgerow networks as wildlife habitat in agricultural landscapes. Pages 111-131 *in* J.R. Park, editor. Environmental management in agriculture-European perspectives. Belhaven Press, London and New York.

Bennett, A.F. 1987. Conservation of mammals within a fragmented environment: the contributions of insular biogeography and autecology. Pages 41-52 *in* D.A. Saunders, G.W. Arnold, A.A. Burbidge, and A.J.M. Hopkins, editors. Nature conservation: the role of remnants of native vegetation. Surrey Beatty and Sons Pry Limited in association with CSIRO and CALM, Sydney.

Bennett, A.F. 1990a. Habitat corridors and the conservation of small mammals in a fragmented forest environment. Landscape Ecology **4(2/3)**: 109-122.

Bennett, A.F. 1990b. Habitat corridors: their role in wildlife management and conservation. Arthur Rylah Institute for Environmental Research. Department of Conservation and Environment, Victoria, Australia.

Brittingham, M.C., and S.A. Temple. 1983. Have cowbirds caused forest songbirds to decline? BioScience **33**: 31-35.

Brown, J.H. 1978. The theory of insular biogeography and the distribution of boreal birds and mammals. The Great Basin Naturalist Memoirs **2**: 210-227.

Burel, F., and J. Baudry. 1990. Hedgerow network patterns and processes in France. Pages 99-120 *in* I.S. Zonneveld and R.T.T. Forman, editors. Changing landscapes: an ecological perspective. Springer-Verlag, New York Inc.

Emmerich, J.M., and P.A. Vohs. 1982. Comparative use of four woodland habitats by birds. Journal of Wildlife Management **46(1)**: 43-49.

Fahrig, L., and G. Merriam. 1985. Habitat patch connectivity and population survival. Ecology **66(6)**: 1762-1768.

Forman, R.T.T., and J. Baudry. 1984. Hedgerows and hedgerow networks in landscape ecology. Environmental Management **8(6)**: 495-510.

Freemark, K.E., and H.G. Merriam. 1986. Importance of area and habitat heterogeneity to bird assemblages in temperate forest fragments. Biological Conservation **36**: 115-141.

Gates, J.E., and L.W. Gysel. 1978. Avian nest dispersion and fledgling success in field-forest ecotones. Ecology **59(5)**: 871-883.

Gottfried, B.M. 1979. Small mammal populations in woodlot islands. American Midland Naturalist **102(1)**: 105-112.

Henderson, M.T., G. Merriam, and J. Wegner. 1985. Patchy environments and species survival: chipmunks in an agricultural mosaic. Biological Conservation **31**: 95-105.

La Polla, V.N., and G.W. Barrett. 1993. Effects of corridor width and presence on the population dynamics of the meadow vole (*Microtus pennsylvanicus*). Landscape Ecology **8(1)**: 25-37.

Lynch, J.F., and D.F. Whigham. 1984. Effects of forest fragmentation on breeding bird communities in Maryland, USA. Biological Conservation **28**: 287-324.

Merriam, G., and A. Lanoue. 1990. Corridor use by small mammals: field measurement for three experimental types of *Peromyscus leucopus*. Landscape Ecology **4(2/3)**: 123-131.

Middleton, J., and G. Merriam. 1983. Distribution of woodland species in farmland woods. Journal of Applied Ecology **20**: 625-644.

Mills, L.S. 1995. Edge effects and isolation: Red-backed voles on forest remnants. Conservation Biology **9(2)**: 395-403.

Noss, R.F. 1983. A regional landscape approach to maintain diversity. BioScience **33(11)**: 700-706.

Pollard, E., and J. Relton. 1970. A study of small mammals in hedges and cultivated fields. Journal of Applied Ecology **7**: 549-557.

Robbins, C.S., J.R. Sauer, R.S. Greenberg, and S. Droege. 1989. Population declines in North American birds that migrate to the neotropics. Proceedings National Academy Science USA **86**: 7658-7662.

Saunders, D.A., R.J. Hobbs, and C.R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology **5(1)**: 18-32.

Stone, D.N.G. 1969. The process of rural settlement in the Athabasca area, Alberta. MA. Thesis, Department of Geography, University of Alberta.

Weatherhill, R.G., and L.B. Keith. 1969. The effect of livestock grazing on an aspen forest community. Department of Lands and Forests Fish and Wildlife Division, Technical Bulletin No. 1.

Wegner, J., and G. Merriam. 1990. Use of spatial elements in a farmland mosaic by a woodland rodent. Biological Conservation **54**: 263-276.

Yahner, R.H. 1983. Small mammals in farmstead shelterbelts: habitat correlates of seasonal abundance and community structure. Journal of Wildlife Management **47(1)**: 74-84.

Zhang, Z., and M.B. Usher. 1991. Dispersal of wood mice and bank voles in an agricultural landscape. Acta Theriologica **36(3/4)**: 239-245.

CHAPTER 2

Aspen fencerows as habitat for small mammals in an agricultural landscape in north central Alberta

2.0 INTRODUCTION

Habitat fragmentation results in loss of original habitat, increased isolation of remaining habitat remnants, and the creation of edge (Saunders *et al.* 1991). Forest fragmentation by agriculture gradually reduces wooded habitat to small woodlots and linear strips of vegetation along field edges (fencerows) isolated from each other by a matrix of crops. This habitat loss and isolation can have a substantial impact on species that are dependent upon wooded habitat (Wegner & Merriam 1990). Local extinctions of populations are frequent in fragmented landscapes (den Boer 1981, Merriam & Wegner 1992) and are affected by woodlot and population size (Opdam *et al.* 1993).

In agricultural landscapes the area of woodlots and linear strips is positively associated with mammalian species richness (Brown 1978, Gottfried 1979, Matthiae & Stearns 1981, Bennett 1987, Yahner 1983a, Bennett 1990), abundance (Matthiae & Stearns 1981, Yahner 1983a, Mills 1995) and probability of occurrence (Smith 1974, Verboom & van Apeldoorn 1990, Celada *et al.* 1994). In areas where fragmentation was extensive, small mammal abundance was highest in the remaining woodlots (Angelstam *et al.* 1987, Yahner 1992) and linear strips (Yahner 1983a & b). In an agricultural landscape fencerows may represent important habitat to small mammal species restricted to wooded habitat (Yahner 1983a) and provide permanent habitat (Wegner & Merriam 1979, Henderson *et al.* 1985, Merriam 1990, Zhang & Usher 1991), with species richness and abundance increasing with vegetation complexity and microhabitat heterogeneity (Yahner 1983a, Forman & Baudry 1984).

The probability of a woodlot being recolonized by immigrants decreases with increased isolation from other available habitat (Howe *et al.* 1991). In addition, in agricultural landscapes small mammal movements between habitats may be further inhibited by crops (Swihart & Slade 1984, Henderson *et al.* 1985). These negative effects of isolation may be reduced by landscape elements that connect woodlots. Fencerows facilitated movement of small mammals between woodlots (Wegner & Merriam 1979, Henderson *et al.* 1985, Bennett 1990) and increased the probability of recolonization when local extinctions occurred (Henderson *et al.* 1985). Crops may also function as habitat for some small mammal species with densities differing among crop types (Wegner & Merriam 1990). The use of crops as habitat and for dispersal may depend upon the cover and resources provided (Zhang & Usher 1991, van Apeldoorn *et al.* 1992) and be influenced by the amount of wooded habitat available (Wegner & Merriam 1990).

The purpose of this study was to determine the relationship between small mammal abundance in fencerows and the fencerow and surrounding landscape characteristics. The study was conducted in an agricultural region dominated by hay, small grains (primarily oats and barley), canola, alfalfa and pasture, interspersed with aspen-dominated boreal mixed-wood woodlots and fencerows. The objectives of this study were to (1) determine the effect of fencerow isolation (isolated/connected to a woodlot) and surrounding landscape type (agricultural/forested) on small mammal abundance, and (2) determine the relationship between small mammal abundance and

fencerow dimensions, vegetation, adjacent land use and surrounding landscape context (amount of wooded habitat measured at two scales).

2.1 METHODS

2.1.1 Study area

Thirty fencerows in a predominantly agricultural region surrounding the Meanook Biological Research Station (54° 37' N, 113° 20' W), near Athabasca, Alberta, Canada, were studied during the summers of 1993 and 1994. The fencerows were approximately 800m long and ranged in width from 5-17m. Fencerows were dominated by trembling aspen (*Populus tremuloides*) and balsam poplar (*P. balsamifera*). Understory vegetation was dominated by rose (*Rosa spp.*), raspberry (*Rubus idaeus*), red-osier dogwood (*Cornus stulonifera*) and saskatoon (*Amelanchier alnifolia*). I chose two landscape types based on the percentage forest cover in a 125km^2 area: (1) agricultural landscape with 20% forest cover, and (2) forested landscape with 48% forest cover. Within each landscape type, 6 fencerows were connected to a woodlot and 9 fencerows were isolated (no connection to a woodlot, range 80-829m from nearest woodlot).

2.1.2 Trapping

A 500m long section in the centre of each fencerow was sampled at 50m intervals for small mammals. Pitfall traps were located along a medial line transect at every 50m station. The traps consisted of 4" diameter polyvinyl chloride (PVC) piping, 15" deep with a square PVC base. Traps were closed with a wooden lid and heavy weight between trapping sessions. The area around each trap was prebaited with sunflower seeds and rolled oats 5 days prior to trapping. Each trap contained food and cotton bedding to

ensure overnight survival of any captured animals. Species, sex, age, weight and reproductive condition of captured animals were recorded and they were then eartagged (National band and tag company, size 1 monel) prior to release.

Six fencerows (3 in each landscape type) were trapped simultaneously for two consecutive nights, and this was repeated until all fencerows were sampled, such that a total trapping session lasted 10 days. Trapping sessions were conducted in July and September in 1993 and in May, July and August in 1994. For each two day trapping session, abundance was calculated as the total number of different individuals captured over the two days.

2.1.3 Vegetation

Vegetation was sampled using the point-quarter method (Cottam & Curtis 1956) every 20m along a medial transect within each fencerow. Shrub and tree density and species composition were determined using the 'Quarter Program' (Krebs 1989). Canopy cover and average tree canopy height were recorded using a spherical densiometer and clinometer respectively. Percent ground cover, leaf litter depth and percent cover, average herbaceous height and percent cover were recorded within a 1m x 20cm quadrat. Down woody material (DWM) was recorded along a 5m transect extending from each vegetation sampling station in the direction of the sampling transect (adapted from Lee *et al.* 1995). The frequency of all DWM touching the transect was recorded using the following diameter categories: A) 0-1.9cm, B) 2-4.9cm, C) 5-10.9 cm, D) >11cm.

2.1.4 Fencerow and landscape characteristics

Landscape and fencerow characteristics were recorded from 1:30,000 aerial photographs taken in 1985 using a Summagraphics SummaSketch digitiser (Model MM1201) and Sigma-Scan (1988). Information was referenced against 1:25,000 photographs taken in 1992, and verified in the field to provide current details. The following fencerow and landscape characteristics were recorded from the photographs:

1) Area (ha) of the fencerow

2) Area (ha) of wooded habitat in a rectangular area $2km^2$ around the fencerow

3) Area (ha) of wooded habitat in a rectangular area $15km^2$ around the fencerow

Crop types in the adjacent fields were recorded.

2.1.5 Statistical analyses

All analyses were performed using SYSTAT (1992) unless otherwise stated. All two-way Kruskal-Wallis analyses were performed by hand.

2.1.5.1 Temporal comparisons

The July trapping session was conducted in both years and abundance data per fencerow were compared using analysis of variance (ANOVA) (*P. maniculatus* and *C. gapperi*) and Kruskal-Wallis (*M. pennsylvanicus* and *Sorex* spp.). Abundance data per fencerow were compared between trapping sessions within each year and between September 1993 and May 1994 with ANOVA and Scheffés post-hoc multiple comparisons (*P. maniculatus* and *C. gapperi*) or Kruskal-Wallis (*M. pennsylvanicus* and *Sorex* spp.). Due to the differences in abundance between trapping sessions (see Results) all further analyses were conducted on individual sessions to determine if different

fencerow and landscape characteristics were influential during different periods of the season.

2.1.5.2 Landscape type and isolation

Landscape type (agricultural/forested) and isolation (isolated/connected) are the two main effects in the study design. Mean abundance data per trapping session were analyzed using two-way unbalanced ANOVA (*P. maniculatus* and *C. gapperi*) and Kruskal-Wallis (*M. pennsylvanicus* and *Sorex* spp). To ensure that any differences were not due to differing vegetation characteristics between the main effects, principal components analysis (PCA)(SAS 1985) was used to summarise the original 15 vegetation variables (Table 2.1). PCA reduces the original data set to a subset of new variables (principal components) which are linear combinations of the original variables (James & McCulloch 1990). The original vegetation variables were not measured on the same scale, therefore the analysis was performed on the correlation matrix to standardize the variables (James & McCulloch 1990). The "root greater than 1" criterion (Kaiser 1958) was used to determine the number of principal components to retain for further analysis. Two-way ANOVAs with the main effects were performed on each of these principal components to determine if there were significant differences in vegetation between isolated/connected fencerows and fencerows in the two landscape types.

2.1.5.3 Fencerow and landscape characteristics

The principal components summarizing the vegetation variables were incorporated with the landscape and fencerow characteristics into either stepwise multiple or logistic regression analyses. Stepwise multiple regression analyses were

performed when there were sufficient abundance data to be normally distributed. In all other circumstances, the abundance data were converted to binary data (presence/absence) and multiple logistic regression analyses performed. To be comparable with the stepwise multiple regression analyses, only main effect logistic regression models were constructed. Stepwise multiple regression is a data exploration technique used to determine a subset of predictor variables which combine parsimony with adequacy of prediction (Sokal & Rohlf 1981). Dependent variables were transformed when necessary to ensure normal distributions. Area variables were square root transformed. For each dependent variable, in all circumstances possible, linearity and homogeneity of variance with each independent variable was achieved. Area of wooded habitat within $2km^2$ (AREA2) and $15km^2$ (AREA15) were correlated which can result in the partial regression coefficients associated with them not reflecting the true dependence of the dependent variable (total abundance/fencerow) on the independent variables, giving spurious results (Zar 1984). Therefore, two different regressions were run, including either AREA2 or AREA15, and the model with the best parameter estimates selected. Stepwise regression was used to relate the vegetation, landscape and fencerow characteristics (Table 2.2) to the total abundance of each species in trapping sessions and logistic regressions were performed in a similar manner to relate the vegetation, landscape and fencerow characteristics to the presence/absence of each species. Residual plots were examined for violations of assumptions and adequacy of the fit of the models.

2.2 RESULTS

2.2.1 Temporal comparisons

A total of 2437 small mammals were captured over two years. Four species dominated the captures: deer mouse, *P. maniculatus* (n=995); red-backed vole, *C. gapperi* (n=1075); meadow vole, *M. pennsylvanicus* (n=98); and shrew, *Sorex spp.* (n=269). Due to the distribution range of shrews, it is probable that *S. cinereus*, masked shrew, dominated captures, however, shrews were not identified rigorously. Captures of the meadow jumping mouse, *Zapus hudsonius* were rare (n=3) and will not be considered further.

P. maniculatus abundance decreased slightly (30%) from July 1993 to July 1994 (Figure 2.1, F=2.58, d.f.=1,58, p=0.114) while all other species increased in abundance (Figure 2.1) with 80.2%, 440% and 208% increases for *C. gapperi* (F=11.23, d.f.=1,58, p=0.001), *M. pennsylvanicus* (H=6.87, d.f=1, p=0.009) and *Sorex spp.* (H=14.925, d.f.=1, p<0.001), respectively.

In 1993, *P. maniculatus* abundance decreased by 60% from July to September (Figure 2.2, F=10.79, d.f.=2, p=0.002). There was a 32% decline in abundance between September 1993 and May 1994 (Figure 2.2, F=8.691, d.f.=1,58, p=0.005). Abundance differed between months in 1994 (Figure 2.2, F=19.37, d.f.=2, p<0.001) with a 448% and 235% increase in abundance from May to July (p<0.001) and May to Aug (p=0.006) respectively, and a 38.8% decrease in abundance from July to August (p=0.016). In 1993, *C. gapperi* increased by 38.9% in abundance between July and September (Figure 2.3, F=3.36, d.f.=1,58, p=0.07). Abundance did not differ from September 1993 to May 1994 (Figure 2.3, F=0.002, d.f.=1,58, p=0.883). In 1994, there was a 38.2% and 23%

increase in abundance for May to July, and May to August respectively, and a 10.7% decrease from July to August (Figure 2.3, F=1.18, d.f.=1, p=0.314). *M. pennsylvanicus* captures were low in all trapping sessions with only 5 captures in both July and September 1993 (Figure 2.4, H=0.14, d.f.=1, p=0.710). There was a 834% increase in abundance from September 1993 to May 1994 (Figure 2.4, H=9.657, d.f.=1, p=0.002). In contrast to the *P. maniculatus* and *C. gapperi* data, the highest abundance in 1994 was in May with a 42.6% and 76.6% decline to July and August respectively, and a 59.3% decrease from July to August (Figure 2.4, H=3.55, d.f.=2, p=0.169). Shrew abundance increased by 50% from July to September in 1993 (Figure 2.5, H=3.404, d.f.=1, p=0.065). Abundance decreased by 82% from September 1993 to May 1994 (Figure 2.5, H=17.55, d.f.=1, p=0.000). In 1994, abundance showed a 1043% increase between May to July, a 1557% from May to August, and a 45% increase from July to August (Figure 2.5, H= 47.65, d.f.=2, p<0.001).

2.2.2 Vegetation

Principal components analysis reduced the vegetation variables to 5 principal components with an eigenvalue >1. These principal components explained 74% of the variance in the original 15 variables (Table 2.1). There was no significant difference in vegetation characteristics between isolated/connected fencerows and the two landscape types for three of the five principal components. One principal component differed significantly between connected and isolated fencerows (F=5.45, d.f.=1,26, p=0.028). There were high positive loadings for herbaceous cover and height, and high negative loadings for the percent canopy cover and DWM in class A. Univariate analysis of these

4 vegetation characteristics indicate that they do not differ significantly between connected and isolated fencerows ($F \leq 2.93$, d.f.=1,26, $p > 0.1$), and it is only when they are combined into a multivariate axis that a difference occurs. There was a significant difference in landscape type for the principal component with high positive loadings for shrub diversity and canopy height, and high negative loadings for leaf depth and DWM in class D ($F=15.34$, d.f.=1,26, $p=0.001$). The univariate analyses of these 4 variables reveal that only shrub diversity and DWM in class D were significantly different between landscape types. Shrub diversity was higher in the forested landscape ($F=9.25$, d.f.=1,26, $p=0.005$) and frequency of DWM in class D increased in the agricultural landscape ($F=10.00$, d.f.=1,26, $p=0.004$).

To adjust for these differences in the two-way ANOVAs with small mammal abundance, these principal components could be used as covariates. However, these principal components did not covary with species abundance and therefore analysis of covariance is unsuitable (Sokal & Rolhf 1981). Vegetation is homogeneous between fencerows for most of the measured characteristics, however, when species abundance between landscape type or connected/isolated differs, it should be kept in mind that these differences may be in part due to the above vegetation characteristics.

2.2.3 Abundance of *Peromyscus maniculatus*

2.2.3.1 Landscape type and isolation

P. maniculatus abundances were higher in the agricultural landscape in all trapping sessions (Figure 2.6), and these differences were significant in July 1993 ($F=11.72$, d.f.=1,26, $p=0.002$), September 1993 ($F=10.1$, d.f.=1,26, $p<0.001$) and July

1994 ($F=5.66$, d.f.=1,26, $p=0.025$). There were no significant differences in abundances between isolated and connected fencerows (Figure 2.7, $F \leq 1.35$, d.f.=1,26, $p>0.05$) and there were no significant interactions between landscape type and isolation ($F \leq 1.55$, d.f.=1,26, $p>0.05$).

2.2.3.2 Fencerow and landscape characteristics

A number of different variables were significant predictors of abundance in the different trapping sessions (Table 2.3) with the total variance explained for different sessions varying between 31.5 and 62.0%. Landscape was the primary predictor for *P. maniculatus* abundance with increased abundance in the agricultural landscape. In July 1993, there was a negative correlation of abundance with area of wooded habitat within 15km^2. In September 1993, there was increased abundance in smaller fencerows with high shrub diversity, low mean litter depth and cover. In addition to landscape type, area and vegetation characteristics were the significant predictors in 1994, however, there is no clear pattern in the effect of vegetation on abundance. For May, logistic regression analysis showed that there was an increased probability of *P. maniculatus* being present in fencerows with decreasing wooded habitat within 2km^2, in connected fencerows and in fencerows with increasing abundance of DWM, and decreasing herbaceous cover and height. In July, abundance increased in the agricultural landscape and in fencerows with increasing abundance of DWM, and decreasing herbaceous cover and height. In August, abundance increased in the agricultural landscape, in smaller fencerows and in those with increasing canopy cover and shrub diversity, but decreasing litter depth and cover, abundance of larger diameter DWM, and shrub density (Table 2.3).

2.2.4 Abundance of *Clethrionomys gapperi*

2.2.4.1 Landscape type and isolation

There was a trend toward higher *C. gapperi* abundance in the forested landscape (Figure 2.8), with these differences being significant in all sessions in 1994 ($F \geq 3.48$, d.f.=1,26, $p \leq 0.05$) but not in 1993 ($F \leq 0.72$, d.f.=1,26, $p > 0.05$). There were no differences in abundance between connected and isolated fencerows in 1993 (Figure 2.9, $F \leq 0.29$, d.f.=1,26, $p > 0.05$) and May 1994 ($F = 0.81$, d.f.=1,26, $p > 0.05$), however, in July and August there were higher abundances in the connected fencerows ($F = 6.23$, d.f.=1,26, $p = 0.05$ and $F = 3.57$, d.f.=1,26, $p > 0.05$, respectively). There were significant interaction effects between landscape type and isolation in May 1994 (Figure 2.10a, $F = 5.269$, d.f.=1,26, $p = 0.03$) and July 1994 (Figure 2.10b, $F = 7.56$, d.f.=1,26, $p = 0.011$) where in the agricultural landscape there were always lower abundances in the connected fencerows than in the isolated, but the reverse was true in the forested landscape.

2.2.4.2 Fencerow and landscape characteristics

Subsets of significant predictors of *C. gapperi* abundance explained between 18.6 and 37.1% of the variance within different sessions (Table 2.4). In July 1993, abundance was higher when there was a greater percent of hay in surrounding fields. In September 1993, higher abundances were found in fencerows with increasing shrub density and litter depth. In 1994, other landscape characteristics were significant predictors. In May, landscape type was the only predictor of abundance with higher abundance in the forested landscape. Landscape type was not an important predictor in July nor August, however, another measure of the area of wooded habitat immediately surrounding the

fencerow (within $2km^2$) was a significant predictor with higher abundances in fencerows with increasing wooded habitat in the immediate vicinity. Additionally, abundances were higher in fencerows with increasing canopy cover and decreasing abundance of larger diameter DWM in July, and with increasing herbaceous height and cover, and decreasing abundance of DWM in August.

2.2.5 Abundance of *Microtus pennsylvanicus*

2.2.5.1 Landscape type and isolation

There was no effect of landscape type (Figure 2.11, $H \leq 1.0$, d.f.=1, $p>0.25$), isolation (Figure 2.12, $H \leq 2.52$, d.f.=1, $p>0.05$) nor interaction between landscape type and isolation on abundance ($H \leq 1.22$, d.f.=1, $p>0.25$).

2.2.5.2 Fencerow and landscape characteristics

Logistic regression analyses were performed on all *M. pennsylvanicus* data (Table 2.5). There were no predictors for the presence/absence data in the 1993 trapping sessions. In May 1994, the probability of *M. pennsylvanicus* being present increased with increasing wooded habitat within $2km^2$. In July and August, the probability of *M. pennsylvanicus* being present increased with increasing wooded habitat within $2km^2$, decreasing fencerow area and decreasing abundance of DWM, and increasing herbaceous cover and height. In addition, the probability of *M. pennsylvanicus* being present increased in connected fencerows and in the forested landscape type in July and August respectively.

2.2.6 Abundance of *Sorex species*

2.2.6.1 Landscape type and isolation

In the July 1993 session there was a higher abundance of shrews in the forested landscape type (Figure 2.13, $H=5.11$, d.f.=1, $p<0.05$), however, in all other sessions the opposite was true (Figure 2.13, $H \leq 1.4$, d.f.=1, $p>0.1$). In 1993 and 1994, there were no differences in abundance between isolated and connected fencerows (Figure 2.14, $H \leq 0.87$, d.f.=1, $p>0.75$). There were no interaction effects between landscape type and isolation in any sessions (Figure 2.14, $H \leq 0.72$, $p>0.75$).

2.2.6.2 Fencerow and landscape characteristics

In three trapping sessions (September, May and August) there were no predictors of shrew presence (Table 2.6). In both July sessions landscape type predicted the presence of shrews, however the results were contradictory with an increased probability of shrews being present in the forested landscape in 1993 and the agricultural landscape in 1994.

2.3 DISCUSSION

P. maniculatus and *C. gapperi* were the most abundant species in this mosaic of widely dispersed wooded habitat and extensive crops. *P. maniculatus* is a woodland specialist (Baker 1968) but is known to inhabit a variety of habitats (Galindo & Krebs 1985), and has been observed in crops (Sadleir 1965, Fleharty & Navo 1983). *C. gapperi* is dependent upon wooded habitat (Yahner 1983a & 1992). *M. pennsylvanicus* and *Sorex species* were captured infrequently. *M. pennsylvanicus* is usually associated with grassland and avoids wooded habitat (Grant 1971). *Sorex* species use a variety of wooded and non-wooded habitat (Iverson *et al.* 1967). *S. cinereus* is a habitat generalist (Iverson

et al. 1967, Yahner 1982 & 1992) and its distribution is unaffected by vegetation, cover or food (e.g. Getz 1961, Wrigley *et al.* 1979).

2.3.1 Landscape type and isolation

The effects of landscape type and isolation were not confounded by differences in vegetation between fencerows in the two landscape types or isolation categories. Landscape type and isolation had opposing effects on *P. maniculatus* and *C. gapperi* abundance. *P. maniculatus* abundance was higher in the agricultural landscape and there was no relationship with isolation. In contrast, at high densities *C. gapperi* abundance were higher in the forested landscape and in connected fencerows.

Landscape type was the predominant variable affecting *P. maniculatus* abundance. Area of wooded habitat within $15km^2$ was negatively associated with abundance in one trapping session, supporting the landscape effect of higher abundance when there was a paucity of available wooded habitat. It appears that when wooded habitat was scarce deer mice were concentrated in the available wooded habitat. I found no relationship between *P. maniculatus* abundance and isolation. Extinctions are frequent in small fragments (Merriam & Wegner 1992) and dispersal among fragments is an important process for the survival of populations (Zhang & Usher 1991). Agricultural fields may represent barriers to dispersal (Mader 1984, Swihart & Slade 1984, Zhang & Usher 1991), resulting in lower immigration rates into isolated fragments and consequently smaller populations with increased chances of extinction (Gottfried 1979). Studies in Ontario on a closely related species, *P. leucopus* (white-footed mouse), have found that the mice use fencerows for movement to relieve the barrier created by fields (Merriam & Lanoue 1990). If fencerows

are acting as conduits I would have expected abundance to be higher in connected fencerows. However, although the isolated fencerows are not directly connected to woodlots they often form part of a network of interconnected fencerows; individuals may be using other fencerows to link up with the fencerows I classified as isolated, therefore these fencerows may not be perceived as isolated by *P. maniculatus*. The increased probability (presence/absence data) of *P. maniculatus* being present in connected fencerows in spring supports the contention that fencerows may be acting as corridors when fields are bare and potentially acting as barriers to movement. Dispersal into a fragment depends upon the barrier between fragments and an individual's ability to overcome this barrier. *P. maniculatus* have been documented in fields (Sadleir 1965, Fleharty & Navo 1983) and therefore the crops may have no inhibitory effect on movement but the ability to use crops for dispersal may be dependent upon the cover provided (Zhang & Usher 1991). Research is needed to investigate if *P. maniculatus* will disperse using the crop types found in this area. An alternative explanation is that the distances between fragments in this study are not enough to influence movements. The distances and patterns of *P. maniculatus* movements need to be investigated in this agricultural landscape.

In the year of higher densities (1994), *C. gapperi* abundance was higher in fencerows in the forested landscape with this trend being seen in the 1993 trapping sessions. This was supported by a positive relationship between abundance and the area of wooded habitat within $2km^2$ of the fencerow in the July and August sessions (1994). In the forested landscape there is a higher proportion of wooded habitat and these nearby

woodlots may serve as source pools for *C. gapperi* recruits to populations in linear strips (Yahner 1983a). These movements may be influenced by the distance between habitats that has to be traversed. I found higher abundance in connected fencerows. *C. glareolus* (bank vole) dispersal into woodlots increased when there were other close woodlots or where woodlots were connected with linear strips (Zhang & Usher 1991). Microtine species rarely cross edges (e.g. Swihart & Slade 1984) and an extension of a habitat by means of a corridor should increase dispersal from that patch (La Polla & Barrett 1993). Studies have shown that small mammals use fencerows as corridors to move between woodlots (Henderson *et al.* 1985, Bennett 1987 & 1990, Merriam & Lanoue 1990). Therefore, *C. gapperi* dispersal would result in higher abundances in fencerows connected directly to a woodlot. This is supported by my results.

There was no effect of landscape type nor isolation on *M. pennsylvanicus* abundance. However, the amount of wooded habitat at a smaller scale surrounding the fencerows influenced the likelihood of a meadow vole being present in a fencerow. *M. pennsylvanicus* were at their highest abundance in fencerows in May and had an increased probability of being found in fencerows with less wooded habitat in the surrounding area. *M. pennsylvanicus* are grassland species and crops are probably their optimal habitat in an agricultural mosaic. During May, fields were bare or had a low amount of cover therefore wooded habitat would be used to a greater extent. When the crops presumably provide the necessary cover and resources (July and August), the opposite relationship is found and is also indicated at the landscape scale with abundance. Areas with more wooded habitat have a lower proportion of crops, increasing the probability of an individual using

alternative habitat e.g. fencerows. Landscape context and isolation did not affect the presence of *S. cinereus* in fencerows. The effect of landscape type was inconclusive.

2.3.2 Fencerow characteristics

There were no consistent relationships between fencerow area and *P. maniculatus*, *C. gapperi* or *S. cinereus* abundances. Similarly, Yahner (1983a) found no relationship between *P. leucopus, C. gapperi* or *S. cinereus* summer abundances and linear strip area. Due to the narrow range in fencerow area in this study (0.44-1.3ha, mean=0.72ha) this result might be expected. The probability of *M. pennsylvanicus* being present increased in smaller fencerows in the latter part of the summer in the year of high abundance. In May, abundances were at their highest and the probability of *M. pennsylvanicus* being present did not differ between fencerows. As previously explained, *M. pennsylvanicus* may use fencerows whilst the fields are bare and subsequently use fields once the crops provide the necessary resources. At this time, meadow voles may encounter fencerows during their daily movements and dispersal and the probability of capturing an individual will be higher in smaller fencerows accounting for the increased likelihood of them being found in smaller fencerows. This is also supported by Iverson *et al.* (1967) who state that wooded habitats are only usually used by *M. pennsylvanicus* transients.

There were no clear patterns in the vegetation variables selected as predictors for *P. maniculatus* abundance in fencerows. Fencerows were chosen to be homogeneous and deer mice were recorded in all fencerows. Studies investigating microhabitat selection have lacked consistency in their results when determining the conditions necessary for *Peromyscus* (Kaufman & Kaufman 1989), perhaps as a consequence of them being able to

live in a variety of habitats. In general, *C. gapperi* abundance increased in fencerows with increasing cover, either overhead canopy cover or ground level cover provided by shrubs and herbs. *C. gapperi* forages at ground level in litter (Yahner 1983a) and this overhead and ground level cover may provide protection from avian and mammalian predators respectively, and the low cover may provide the preferred moist microenvironments (Yahner 1992).

M. pennsylvanicus are associated with grasslands (Iverson *et al.* 1967, Grant 1971, Getz 1985) but may also be found in small, grassy patches within forested areas (Getz 1985). Vegetation is one of the most important factors influencing the presence of *Microtus* species, providing protection, concealment and food (Rose and Birney 1985). The probability of *M. pennsylvanicus* being present in a fencerow increased with increasing herbaceous cover and height which provide dense vegetative cover and food. Vegetation variables generally do not affect *Sorex* species (Wrigley *et al.* 1979). There were no local vegetation characteristics as predictors for the presence of *S. cinereus* in fencerows.

Adjacent land use did not influence *P. maniculatus* abundance. *P. leucopus* inhabit a variety of crops with agricultural intensity influencing the relative use of crops and wooded habitat (Wegner & Merriam 1990). *P. leucopus* densities in corn and small grain crops were comparable to densities in woodlots, but lower in hay fields and pasture (Wegner & Merriam 1990). They enter fields of small grains when heads are forming in May and may stay until harvest in August (Merriam 1990), however, none remain after harvest (Wegner & Merriam 1990). This species changes its relative use of fencerows and

crops according to resource availability through the growing season. *P. maniculatus* may also use crops as habitat providing there is sufficient cover and resources available, however, adjacent land use was not an influential factor suggesting that deer mice are either not using the crops or using all crops equally. *P. maniculatus* needs to be investigated in this region to determine the relative use of woodlots, fencerows and crops as habitat. Use of crops by deer mice may be influenced by the growing season of the crop, the resources available, and agricultural disturbances.

In the year of lower densities, *C. gapperi* abundance was higher in fencerows with a greater proportion of hay (lower proportion of small grains) in the surrounding fields. This result suggests that either hay fields are avoided by *C. gapperi* and fencerows are the optimal habitat in this landscape, or that densities were high in the hay fields resulting in the surplus *C. gapperi* inhabiting fencerows. However, *C. gapperi* are dependent on wooded habitat (Yahner 1983a & 1992), therefore, the former explanation is more probable. There was no effect of crop type during the year of higher abundance; individuals may have been forced to occupy all habitats including hay. It has been shown that during periods of high density individuals may be found temporarily in marginal habitats (Getz 1985, Rose & Birney 1985). There was no effect of adjacent land use on *M. pennsylvanicus* or *S. cinereus* presence.

2.3.3 Demography

Between and within year population fluctuations occurred. *P. maniculatus* populations usually do not undergo interannual fluctuations (Termon 1966, Gilbert & Krebs 1991) while abundance of other rodent species may fluctuate annually due to

density-dependent or density-independent factors (Gilbert & Krebs 1991, Yahner 1992). My results demonstrated this, with *P. maniculatus* showing a slight decline between years while *C. gapperi* increased significantly from 1993 to 1994. There was also a dramatic increase in microtine rodent abundance in study sites approximately 100 km east of this area (R. Moses, pers. comm.), suggesting that the small mammals in the fencerows were undergoing similar fluctuations as populations over a larger geographic area.

Small mammal populations also fluctuate within a breeding season from low numbers in spring (due to overwinter mortality) to peaks in autumn (Terman 1968, Wegner & Merriam 1990, van Apeldoorn *et al.* 1992). *P. maniculatus* abundance in fencerows declined slightly earlier (August/September) in comparison to other studies where populations peak in August/September (e.g. Yahner 1983b, Fahrig & Merriam 1985, Galindo & Krebs 1985) and showed a significant decline between fall and spring. In comparison, *C. gapperi* abundance remained virtually the same between fall and spring, perhaps due to their longer breeding season (Gilbert & Krebs 1991). Both species had low overwinter survival (3%, unpubl. data) indicating complete turnover. The fencerows were therefore recolonized through immigration. All fencerows had unique individuals, therefore, fencerows were not isolated to an extent that immigrants could not reach them. It could be argued that individuals captured are only transients as opposed to residents. However, data for both species indicate that there are resident individuals in the fencerows: (1) individuals were captured in more than one session, (2) pregnant and lactating females were captured, (3) young were born in the area (litters were born in traps

and I used a conservative weight estimate (<10g) to determine any young born in the immediate area (R. Weaver pers. comm.)).

M. pennsylvanicus increased significantly between years and throughout the summer in 1994. *Microtus* species may undergo either annual or cyclic fluctuations (Taitt & Krebs 1985). Annual fluctuations are characterized by autumn/winter peaks and a substantial spring decline, while cyclic peak densities demonstrate no spring decline (Taitt & Krebs 1985). It is difficult to classify which fluctuation type populations are undergoing when studies are short-term (Taitt & Krebs 1985). I lack the autumn/winter sessions necessary to determine if declines occurred and the abundances recorded were low. These factors make it difficult to determine the demographic patterns occurring in this area during the study. *M. pennsylvanicus* increased in abundance from autumn to spring with low overwinter survival, again indicating complete turnover and subsequent recolonization through immigration.

S. cinereus abundance increased throughout the season in both years but declined from autumn to spring. No data are available to determine overwinter survival or the nature (i.e. age/sex/immigrants/residents) of the individuals captured.

2.3.4 Fencerows as habitat

The amount of wooded habitat in the surrounding area was the primary predictor for *P. maniculatus* and *C. gapperi* with higher abundances in the agricultural and forested landscapes respectively. The results indicate that isolation could play a role in small mammal dynamics, however, in this area isolation does not appear to be a limiting factor.

These fencerows are habitat fragments that may provide valuable habitat for small mammals in an agricultural landscape. Even the smallest fencerows contained small mammals, indicating that these small fragments may serve important functions for the persistence of small mammals (Fahrig & Merriam 1994). In addition to their role as habitat, interconnecting fencerows may serve as movement corridors between fragments, increasing connectivity and decreasing the probability of extinctions. Fencerows, and their role in species persistence, need to be examined in conjunction with the other habitats to investigate the dynamics of the system and the scale at which agricultural fragmentation affects small mammals.

Despite a variety of characteristics as predictors, none consistently explained the variation in small mammal species abundance in fencerows. Alternatives that might account for this include:

(1) Variables other than the ones measured affect abundance

(2) Variables are only influential during specific periods

(3) The range of the characteristics measured was not extreme enough e.g. isolation distances were not great enough to affect dispersal in these species. However, this study is representative of the landscape in this area.

(4) The species present in this landscape are adapted to this scale of fragmentation and agricultural disturbance after nearly 100 years of agricultural practices.

(5) The index of small mammal abundance was not sufficient to accurately detect differences between fencerows. The probability of capture may depend upon the behavior

of the small mammals, the area sampled relative to the actual area used by individuals, and the proportion of time spent by individuals in that area (Wegner & Merriam 1990).

Table 2.1: Eigenvectors from the principal components analysis used to summarize the original 15 vegetation variables.

Variable	PC1	PC2	PC3	PC4	PC5
Canopy height	.13	-.24	.36	-.29	.18
% canopy cover	.15	-.37	-.28	.12	-.25
Mean height of herbs	-.36	.20	.29	-.18	.03
Log herb cover	-.38	.23	.20	-.23	.10
Mean litter depth	-.08	0	-.20	.52	.34
% litter cover	-.21	-.23	.38	.38	-.11
Log % rock cover	.30	.27	-.27	-.26	-.05
Shrub density	.07	.16	.03	.03	.80
Shrub diversity	.29	-.24	.10	-.37	-.03
Tree density	-.20	-.12	-.37	-.06	.14
Tree diversity	-.01	-.29	.24	.25	-.26
DWM class A	.45	-.07	.04	.06	.12
DWM class B	.34	.19	.36	.09	.09
DWM class C	.32	.31	.25	.27	-.05
DWM class D	.07	.52	-.09	.21	-.04
% variance explained	22	18	16	10	8

Table 2.2: The fencerow, landscape and vegetation predictor variables entered in stepwise multiple regression analyses of the trapping session data (total abundance per fencerow).

1) Landscape type (agricultural/forested) coded as a dummy variable (LAND)
2) Isolated/connected coded as a dummy variable (ISOL)
3) Area (ha) of the fencerow (AREA)
4) Area (ha) of wooded habitat in a rectangular area 2km^2 around the fencerow (AREA2)
5) Area (ha) of wooded habitat in a rectangular area 15km^2 around the fencerow (AREA15)
6) Adjacent crops that were hay, alfalfa and pasture were combined into a hay category and crops of oats and barley were combined into a grain category. The two categories encompassed the crops adjacent to the fencerows and were inversely related. They were represented as one category where the percent of hay was coded as dummy variables (HAY1 and HAY2)
7) Principal Component 1
8) Principal Component 2
9) Principal Component 3
10) Principal Component 4
11) Principal Component 5

Table 2.3: Stepwise regression analyses for *P. maniculatus* trapping session data (total abundance per fencerow). Subsets of predictor variables for each trapping session and the variance explained are shown. Logistic regression analysis for May trapping session with the predictors for increased probability of *P. maniculatus* being present are shown.

Dependent variable	Independent variables	Total variance(%)
July 1993	(1) LAND[1] (2) AREA15(-)	37.8
Sept 1993	(1) LAND[1] (2) AREA(-) (3) PC4(-)	55.6
May 1994	(1) AREA2(-) (2) CONN[1] (3) PC1(+)	
July 1994	(1) LAND[1] (2) PC1(+)	31.5
Aug 1994	(1) LAND[1] (2) AREA(-) (3) PC2(-) (4) PC4(-) (5) PC5(-)	62.0

Categorical variables:
LAND[1]: Higher abundance in agricultural landscape
CONN[1]: Increased probability of *P. maniculatus* being present in a connected fencerow

Table 2.4: Stepwise regression analyses for *C. gapperi* trapping session data (total abundance per fencerow). Subsets of predictor variables for each trapping session and the variance explained are shown.

Dependent variable	**Independent variables**	**Total variance(%)**
July 1993	(1) HAY1[1]	18.6
Sept 1993	(1) HAY1[1] (2) PC5(+)	19.5
May 1994	(1) LAND[2]	15.8
July 1994	(1) AREA2(+)(2) PC2(-)	37.1
Aug 1994	(1) AREA2(+)(2) PC1(-) (3) PC2(-)	36.9

Categorical variables:
HAY1[1]: Higher abundance with high % hay (low % grain)
LAND[2]: Higher abundance in forested landscape

Table 2.5: Logistic regression analyses of presence/absence data for *M. pennsylvanicus*. Predictor variables for the increased probability of *M. pennsylvanicus* being present are shown for each trapping session.

Dependent variable	**Independent variables**
July 1993	No predictor variables
September 1993	No predictor variables
May 1994	(1) AREA2(-)
July 1994	(1) AREA2(+) (2) AREA(-) (3) CONN[1] (4) PC1(-)
August 1994	(1) AREA2(+) (2) AREA(-) (3) LAND[1] (4) PC1(-)

Categorical variables:
CONN[1]: Increased probability of *M. pennsylvanicus* being present in connected fencerows
LAND[1]: Increased probability of *M. pennsylvanicus* being present in forested landscape type

Table 2.6: Logistic regression analyses of presence/absence data for *Sorex spp.* Predictor variables for the increased probability of *Sorex spp.* being present are shown for each trapping session.

Dependent variable	**Independent variables**
July 1993	(1) LAND[1]
September 1993	No predictor variables
May 1994	No predictor variables
July 1994	(1) LAND[2]
August 1994	No predictor variables

Categorical variables:
LAND[1]: Increased probability of *Sorex spp.* being present in forested landscape type
LAND[2]: Increased probability of *Sorex spp.* being present in agricultural landscape type

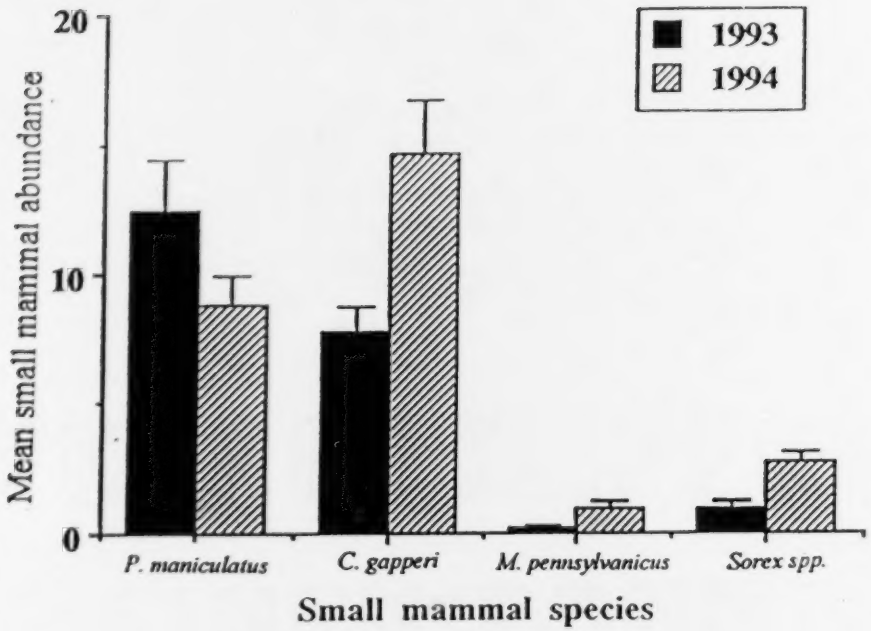

Figure 2.1: Mean small mammal abundance in July 1993 and July 1994; bars represent standard error.

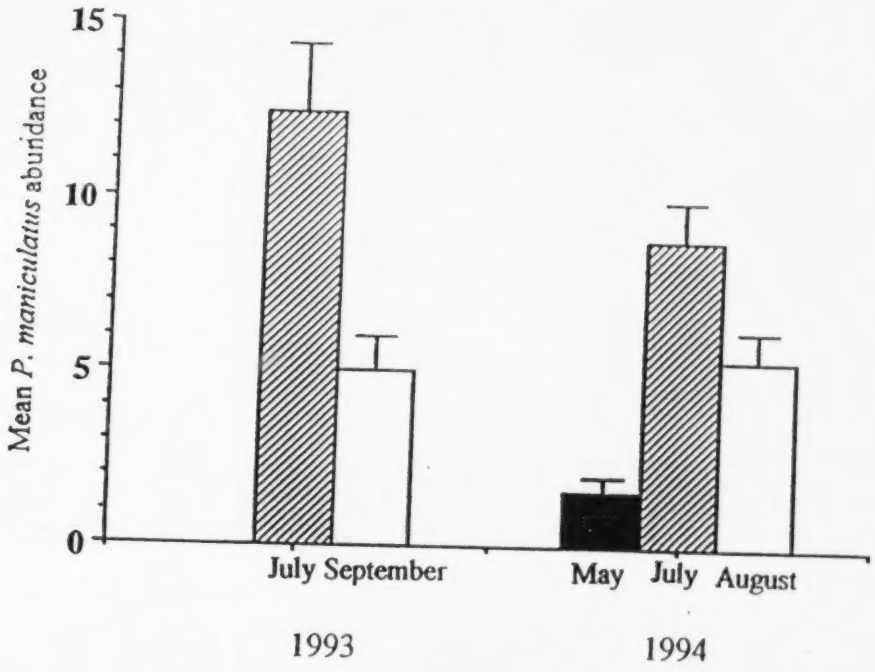

Figure 2.2: Mean *P. maniculatus* abundance for trapping sessions in 1993 and 1994; bars represent standard error.

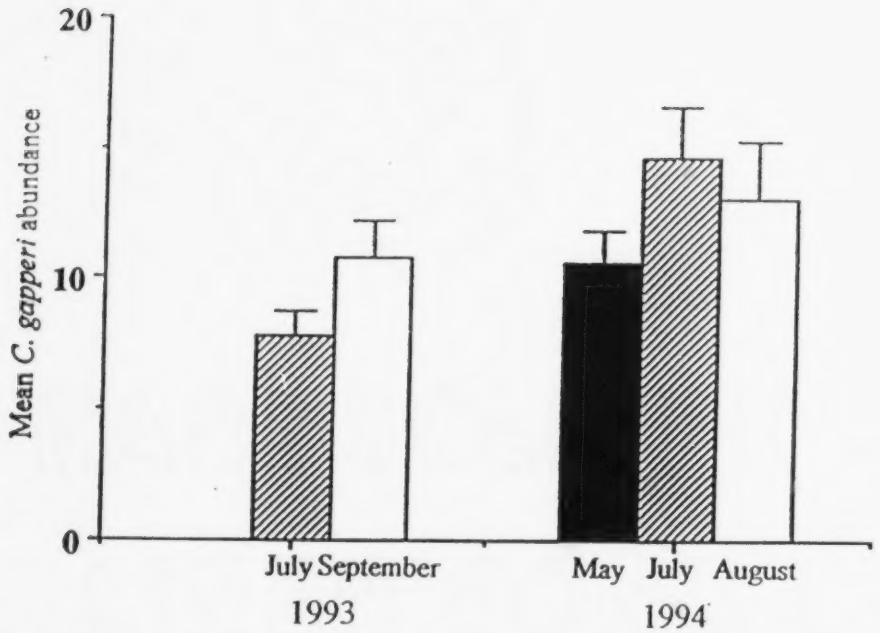

Figure 2.3: Mean *C. gapperi* abundance for trapping sessions in 1993 and 1994; bars represent standard error.

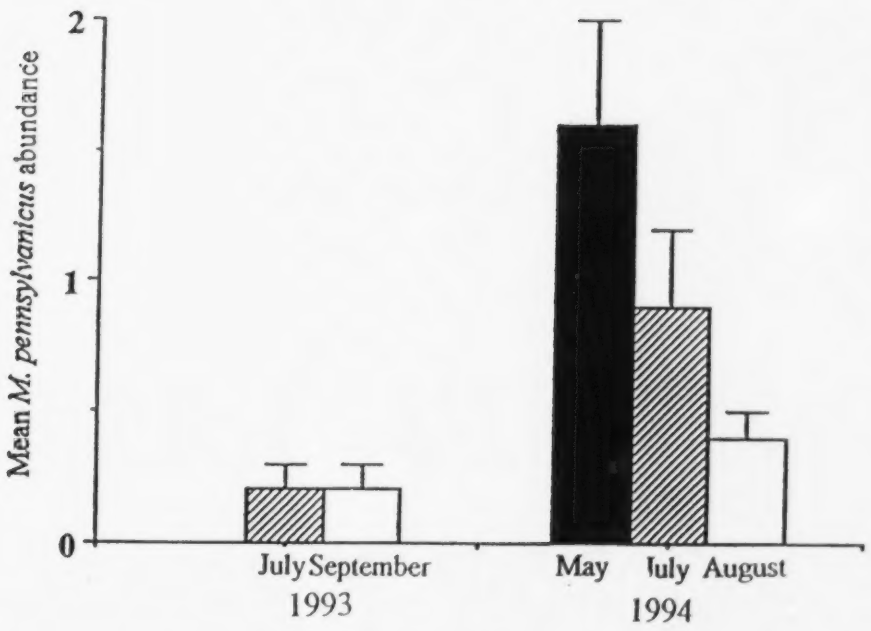

Figure 2.4: Mean *M. pennsylvanicus* abundance for trapping sessions in 1993 and 1994; bars represent standard error.

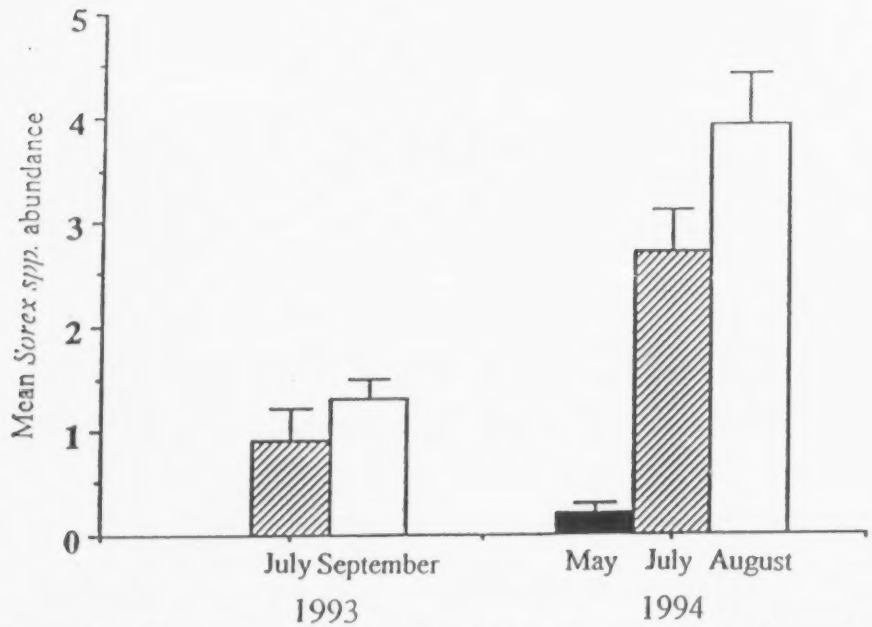

Figure 2.5: Mean *Sorex spp.* abundance for trapping sessions in 1993 and 1994; bars represent standard error.

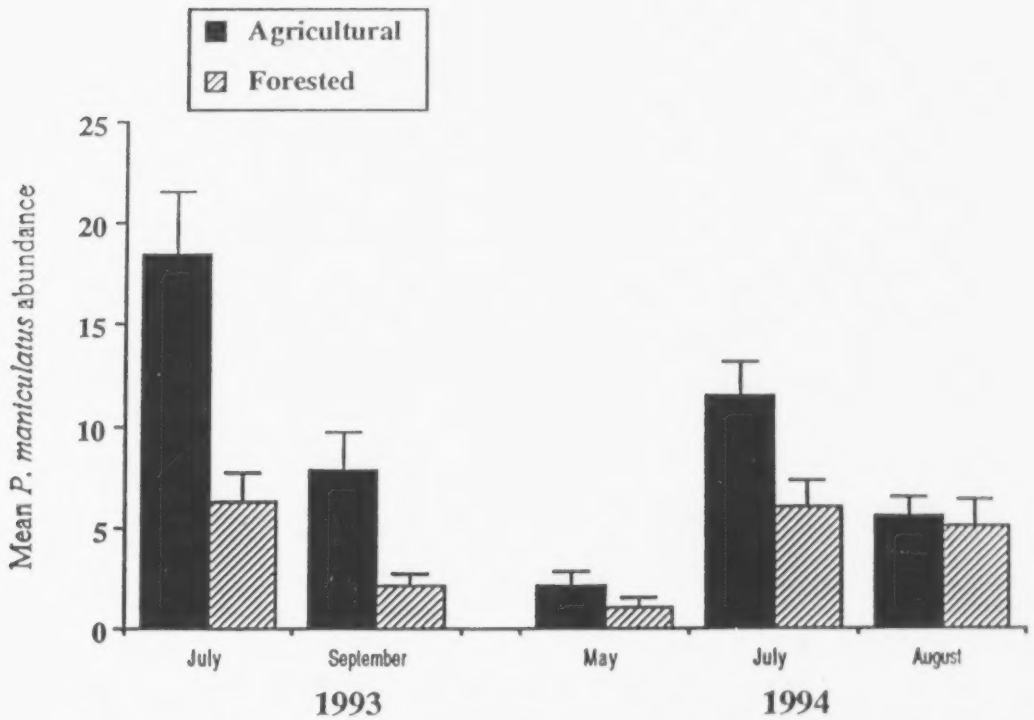

Figure 2.6: Mean *P. maniculatus* abundance in agricultural (n=15) and forested (n=15) landscape types; bars represent standard error.

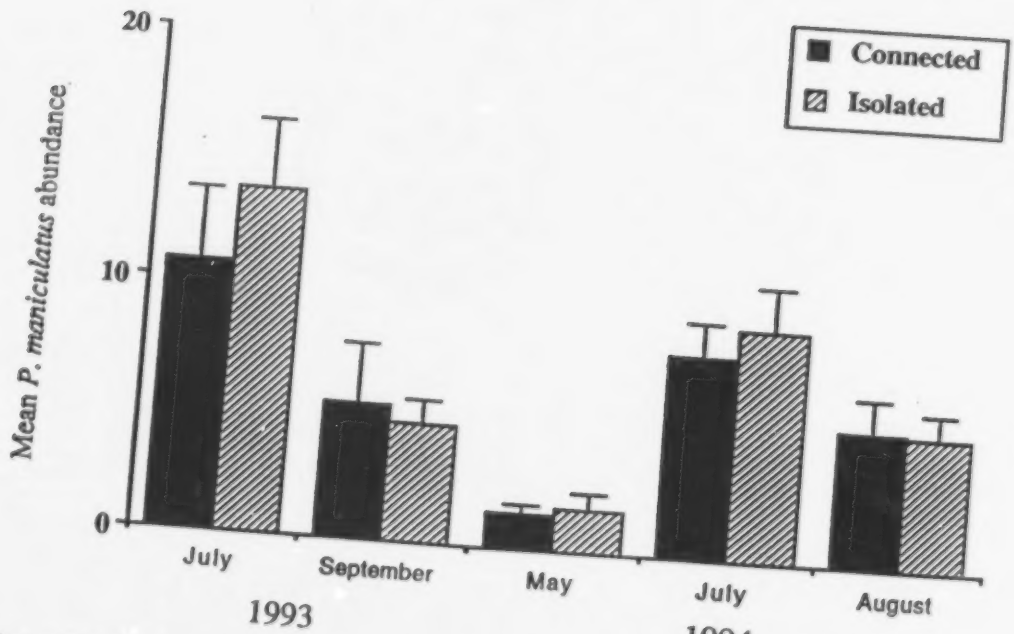

Figure 2.7: Mean *P. maniculatus* abundance in connected (n=12) and isolated (n=18) fencerows; bars represent standard error.

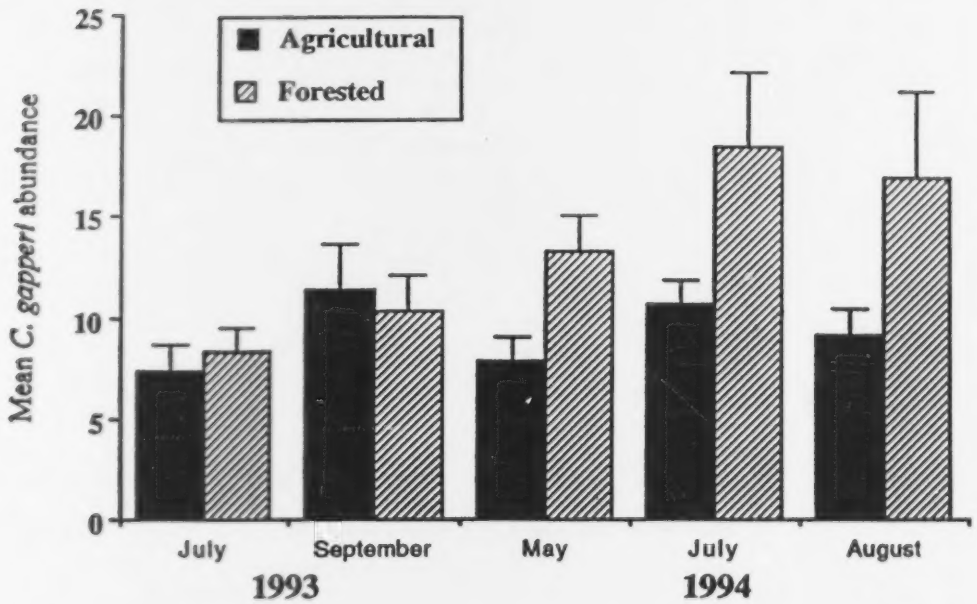

Figure 2.8: Mean *C. gapperi* abundance in agricultural (n=15) and forested (n=15) landscape types; bars represent standard error.

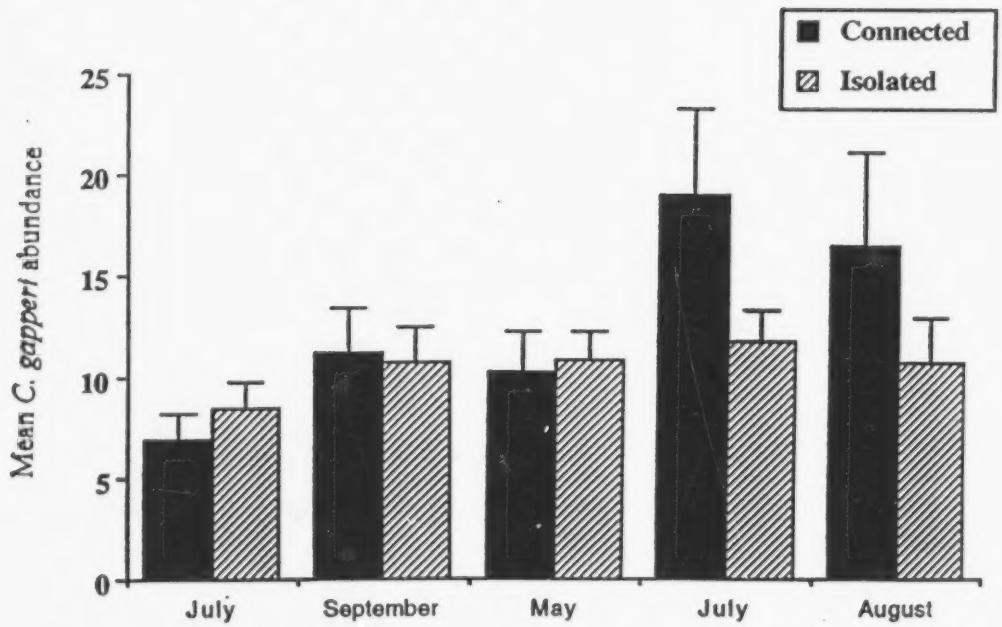

Figure 2.9: Mean *C. gapperi* abundance in connected (n=12) and isolated (n=18) fencerows; bars represent standard error

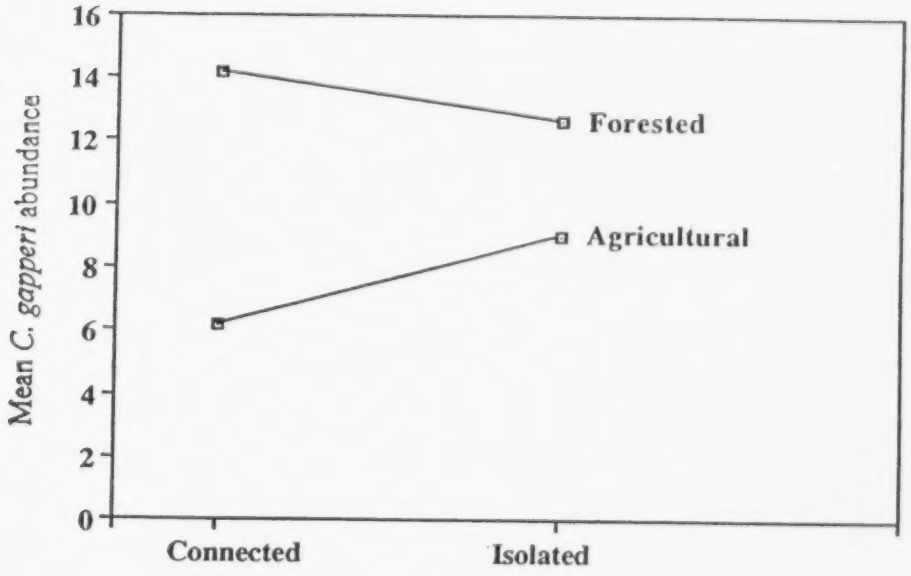

Figure 2.10a: Interaction plot between landscape type (agricultural/forested) and isolation (isolated/connected) for mean *C. gapperi* abundance in the May 1994 trapping session.

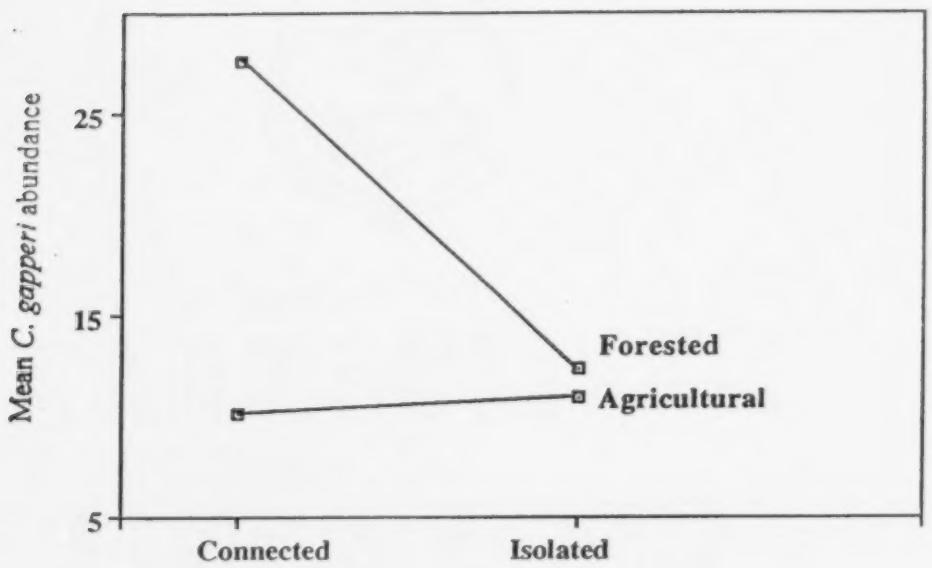

Figure 2.10b: Interaction plot between landscape type (agricultural/forested) and isolation (isolated/connected) for mean *C. gapperi* abundance in the July 1994 trapping session.

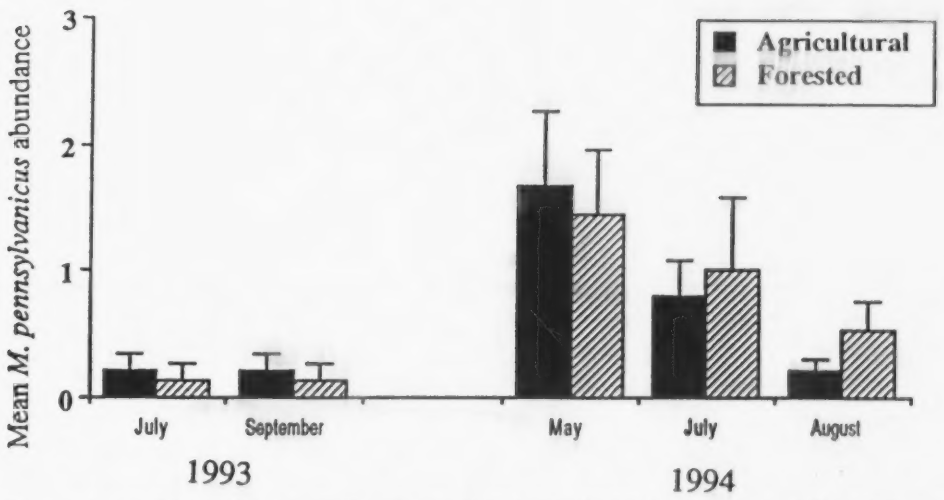

Figure 2.11: Mean *M. pennsylvanicus* abundance in agricultural (n=15) and forested (n=15) landscape types; bars represent standard error.

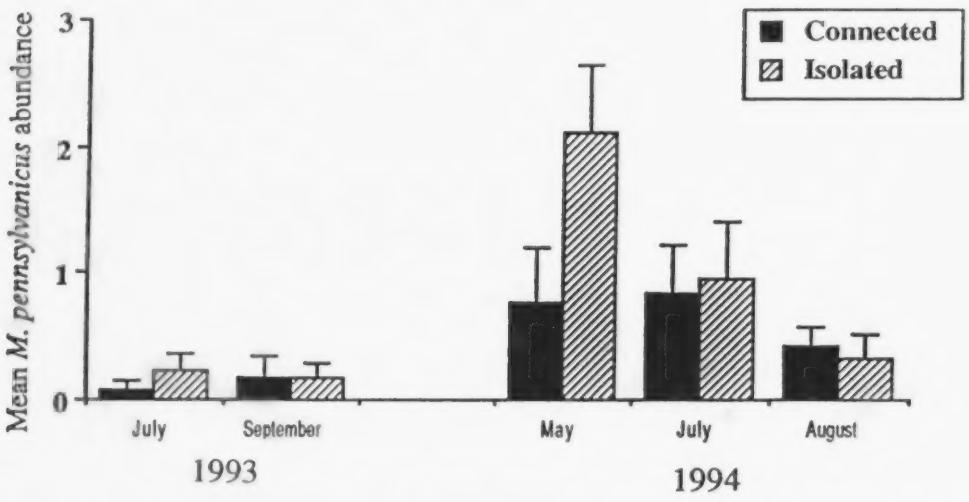

Figure 2.12: Mean *M. pennsylvanicus* abundance in connected (n=12) and isolated (n=18) fencerows; bars represent standard error.

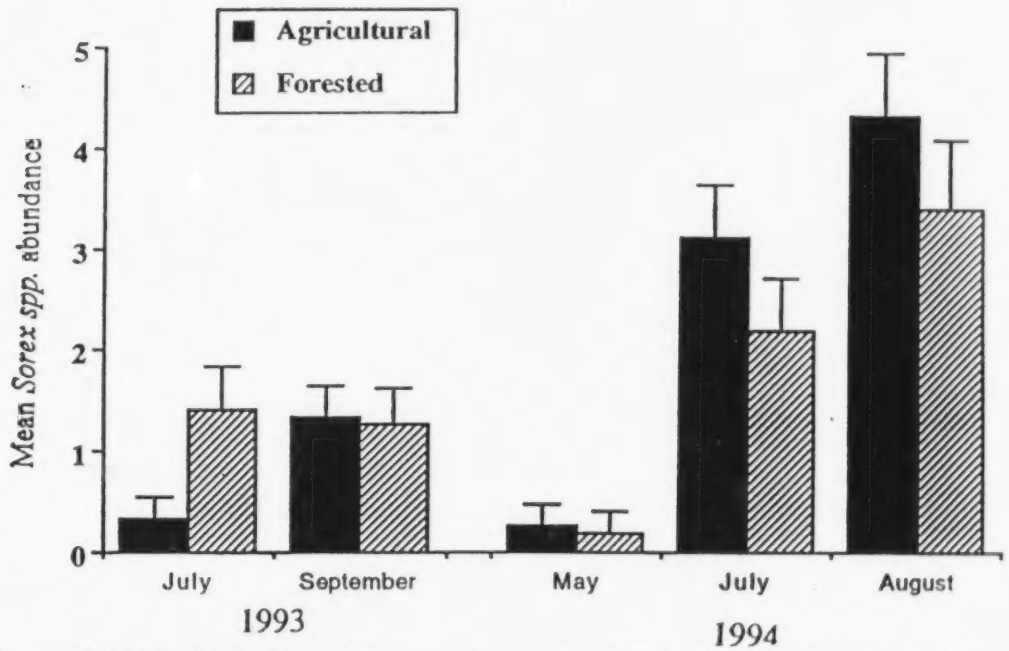

Figure 2.13: Mean *Sorex spp.* abundance in agricultural (n=15) and forested (n=15) landscape types; bars represent standard error.

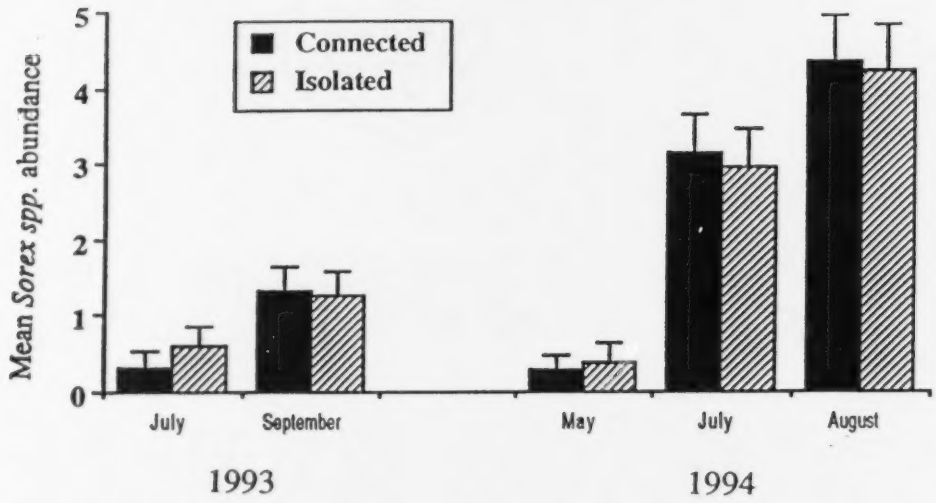

Figure 2.14: Mean *Sorex spp.* abundance in connected (n=12) and isolated (n=18) fencerows; bars represent standard error.

2.4 LITERATURE CITED

Angelstam, P., L. Hansson, and S. Pehrsson. 1987. Distribution borders of field mice *Apodemus*: the importance of seed abundance and landscape composition. Oikos **50**: 123-130.

Baker, R.H. 1968. Habitats and distribution. Pages 98-126 *in* J.A. King, editor. Biology of *Peromyscus* (Rodentia). Special publication No. 2. The American Society of Mammalogists.

Bennett, A.F. 1987. Conservation of mammals within a fragmented forest environment: the contributions of insular biogeography and autecology. Pages 41-52 *in* D.A. Saunders, G.W. Arnold, A.A. Burbidge, and A.J.M. Hopkins, editors. Nature conservation: the role of remnants of native vegetation. Surrey Beatty and Sons Pry Limited in association with CSIRO and CALM, Sydney.

Bennett, A.F. 1990. Habitat corridors and the conservation of small mammals in a fragmented forest environment. Landscape Ecology **4(2/3)**: 109-122.

Brown, J.H. 1978. The theory of insular biogeography and the distribution of boreal birds and mammals. Great Basin Naturalist Memoirs **2**: 209-227.

Celada, C., G. Bogliani, A. Ggariboldi, and A. Maracci. 1994. Occupancy of isolated woodlots by the red squirrel *Sciurus vulgaris* L. in Italy. Biological Conservation **69**: 177-183.

Cottam, G., and J.T. Curtis. 1956. The use of distance methods in phytosociological sampling. Ecology **37**: 451-460.

den Boer, P.J. 1981. On the survival of populations in a heterogeneous and variable environment. Oecologia **50**: 39-53

Fahrig, L. and G. Merriam. 1985. Habitat patch connectivity and population survival. Ecology **66(6):** 1762-1768.

Fahrig, L. and G. Merriam. 1994. Conservation of fragmented populations. Conservation Biology **8(1):** 50-59.

Fleharty, E.D., and K.W. Navo. 1983. Irrigated cornfields as habitat for small mammals in the Sandsage Praire region of western Kansas. Journal of Mammalogy **64**: 367-379.

Forman, R.T.T., and J. Baudry. 1984. Hedgerows and hedgerow networks in landscape ecology. Environmental Management 8(6): 495-510.

Galindo, C., and C.J. Krebs. 1985. Habitat use and abundance of deer mice: interactions with meadow voles and red-backed voles. Canadian Journal of Zoology **63**: 1870-1879.

Getz, L.L. 1961. Factors influencing the local distribution of *Microtus* and *Synaptomys* in southern Michigan. Ecology **41**: 110-119.

Getz, L.L. 1985. Habitats. Pages 286-309 *in* R.H. Tamarin, editor. Biology of New World *Microtus.* Special Publication No. 8. The American Society of Mammologists.

Gilbert, B.S., and C.J. Krebs. 1991. Population dynamics of *Clethrionomys* and *Peromyscus* in southwestern Yukon 1973-1989. Holartic Ecology **14**: 250-259.

Gottfried, B.M. 1979. Small mammal populations in woodlot islands. American Midland Naturalist **102(1)**: 103-112.

Grant, P.R. 1971. The habitat preference of *Microtus pennsylvanicus,* and its relevance to the distribution of this species on islands. Journal of Mammology **52**: 351-361.

Henderson, M.T., G. Merriam, and J. Wegner. 1985. Patchy environments and species survival: chipmunks in an agricultural mosaic. Biological Conservation **31**: 95-105.

Howe, R.W, G.J. Davis, and V. Mosca. 1991. The demographic significance of 'sink' populations. Biological Conservation. 57: 239-255.

Iverson, S.L., R.W. Seabloom, and J.M. Hnatiuk. 1967. Small-mammal distributions across the praire-forest transition of Minnesota and North Dakota. The American Midland Naturalist **78(1):** 188-197.

James, F.C., and C.E. McCulloch. 1990. Multivariate analysis in ecology and systematics: Panacea or Pandora's Box? Annual Review of Ecology and Systematics 21: 129-166.

Kaiser H.F. 1958. The varimax criterion for analytic rotation in factor analysis. Psychometrika **23**: 187-200.

Kaufman, D.W., and G.A. Kaufman. 1989. Population Biology. Pages 233-270 *in* G.L. Kirkland Jr., and J.N. Layne, editors. Advances in the study of *Peromyscus* (Rodentia). Texas Tech University Press, Texas, USA.

Krebs, C.J. 1989. Ecological Methodology. Harper Collins Publishers, New York. 654pp.

La Polla, V.N., and G.W. Barrett. 1993. Effects of corridor width and presence on the population dynamics of the meadow vole (*Microtus pennsylvanicus*). Landscape Ecology **8(1)**: 25-37.

Lee P.C., S. Crites, and J.B. Stelfox. 1995. Changes in forest structure and floral composition in a chronosequence of aspen mixedwood stands in Alberta. Pages 29-48 *in* J.B. Stelfox, editor. Relationships between stand age, stand structure, and biodiversity in aspen mixedwood forests in Alberta. Jointly published by Alberta Environmental Centre (AECV95-R1), Vegreville, AB, and Canadian Forest Service (Project No. 0001A), Edmonton, AB.

Mader, H.J. 1984. Animal habitat isolation by roads and agricultural fields. Biological Conservation **29**: 81-96.

Matthiae, P.E., and F. Stearns. 1981. Mammals in forest islands in southeastern Wisconsin. Pages 55-66 *in* R.L. Burgess and D.M. Sharpe, editors. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, Inc.

Merriam, G. 1990. Ecological processes in the time and space of farmland mosaics. Pages 121-133. *in* I.S. Zonneveld and R.T.T. Forman, editors. Changing landscapes: an ecological perspective. Springer-Verlag, New York, Inc.

Merriam, G., and A. Lanoue. 1990. Corridor use by small mammals: field measurements for three experimental types of *Peromyscus leucopus*. Landscape Ecology **4**: 123-31.

Merriam, G., and J. Wegner. 1992. Local extinctions, habitat fragmentation, and ecotones. Pages 150-169 *in* A.J. Hansen and F. di Castri, editors. Landscape boundaries: consequences for biotic diversity and ecological flows. Springer-Verlag, New York, Inc.

Mills, L.S. 1995. Edge effects and isolation: red-backed voles on forest remnants. Conservation Biology **9(2)**: 395-403.

Opdam, P., R. van Apeldoorn, A. Schotman, and J. Kalkhoven. 1993. Population responses to landscape fragmentation. Pages 147-171 *in* C.C. Vos and P. Opdam, editors. Landscape ecology of a stressed environment. Chapman and Hall, London.

Rose, R.K., and E.C. Birney. 1985. Community ecology. Pages 310-339 *in* R.H. Tamarin, editor. Biology of New World *Microtus*. Special publication No. 8. The American Society of Mammalogists.

Sadleir, R.M.F.S. 1965. The relationship between agonistic behaviour and population changes in the deer mouse (*Peromyscus maniculatus*). Journal of Animal Ecology **34**: 331-352.

SAS Institute Inc. SAS User's Guide: Basics, Version 5 Edition. Cary, NC: SAS Institute Inc., 1985. 1290pp.

Saunders, D.A., R.J. Hobbs, and C.R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5(1): 18-32.

Sigma-Scan V3.90. 1988. The Scientific Measurement Program. Jandel Scientific.

Smith, A.T. 1974. The distribution and dispersal of pikas: consequences of insular population structure. Ecology **55**: 1112-1119.

Sokal, R.R., and Rohlf, F.J. 1981. Biometry. 2nd Edition. W.H. Freeman and Company, New York. 859pp.

Swihart, R.K., and N.A. Slade. 1984. Road crossing in *Sigmodon hispidus* and *Microtus ochrogaster*. Journal of Mammalogy **65(2)**: 357-360.

SYSTAT for windows: statistics, version 5 edition. Evanston IL:SYSTAT, Inc., 1992.

Taitt, M.J., and C.J. Krebs. 1985. Population dynamics and cycles. Pages 567-620 *in* R.H. Tamarin, editor. Biology of New World *Microtus*. Special publication No. 8. The American Society of Mammalogists.

Terman, C.R. 1966. Population fluctuations of *Peromyscus maniculatus* and other small mammals as revealed by the North American census of small mammals. American Midland Naturalist **76**: 419-426.

Terman, C.R. 1968. Population dynamics. Pages 412-450 *in* J.A. King, editor. Biology of *Peromyscus* (Rodentia). Special Publication No. 2. The American Society of Mammologists.

van Apeldoorn, R.C., W.T. Oostenbrink, A. van Winden, and F.F. van der Zee. 1992. Effects of habitat fragmentation on the bank vole, *Clethrionomys glareolus*, in an agricultural landscape. Oikos **65**: 265-274.

Verboom, B., and R. van Apeldoorn. 1990. Effects of habitat fragmentation on the red squirrel, *Sciurus vulgaris* L. Landscape Ecology **4(2/3)**: 171-176.

Wegner, J.F., and G. Merriam. 1979. Movements by birds and small mammals between a wood and adjoining farmland habitats. Journal of Applied Ecology **16**: 349-357.

Wegner, J.F., and G. Merriam. 1990. Use of spatial elements in a farmland mosaic by a woodland rodent. Biological Conservation 54: 263-276.

Wrigley, R.E., J.E. Dubois, and H.W.R. Copeland. 1979. Habitat, abundance, and distribution of six species of shrews in Manitoba. Journal of Mammalogy **60**: 505-520.

Yahner, R.H. 1982. Microhabitat use by small mammals in farmstead shelterbelts. Journal of Mammalogy **63(3)**: 440-445.

Yahner, R.H. 1983a. Small mammals in farmstead shelterbelts: habitat correlates of seasonal abundance and community structure. Journal of Wildlife Management **47(1)**: 74-84.

Yahner, R.H. 1983b. Population dynamics of small mammals in farmstead shelterbelts. Journal of Mammalogy **64(3)**: 380-386.

Yahner, R.H. 1992. Dynamics of a small mammal community in a fragmented forest. American Midland Naturalist **127**: 381-391.

Zar, J.H. 1984. Biostatistical analysis. Second Edition. Prentice Hall, New Jersey. 718pp.

Zhang, Z., and M.B. Usher. 1991. Dispersal of wood mice and bank voles in an agricultural landscape. Acta Theriologica **36(3/4)**: 239-245.

CHAPTER 3

Aspen fencerows as habitat for birds in an agricultural landscape in north central Alberta

3.0 INTRODUCTION

Declines in bird species, especially neotropical migrants, have been partly attributed to fragmentation of wooded habitat (Whitcomb *et al.* 1981, Robbins *et al.* 1989). In agricultural landscapes, forested areas remain as small woodlots or fencerows (linear strips of vegetation along field edges). Fencerows may be important in connecting forest patches (e.g. Wegner & Merriam 1979, Henderson *et al.* 1985, Bennett 1990) but they could also be important as habitat in their own right. There has been little research on the importance of fencerows to bird populations in fragmented landscapes (but see Yahner 1982 & 1983, Shalaway 1985).

In an agriculturally fragmented landscape with few woodlots, fencerows may be critical to the survival of many species (Forman & Baudry 1984). However, fencerows are essentially small fragments with correspondingly small populations which have increased probabilities of extinction through stochastic processes (Askins *et al.* 1987, Freemark & Merriam 1989). Fencerows may be smaller than the minimum territory size of some species, not include enough territories to maintain a viable population or lack the microhabitats necessary for many species (Askins *et al.* 1987). Very small fragments may be entirely impacted by negative edge effects, and may function as ecological traps for many species due to increased competition, predation and brood parasitism (Brittingham & Temple 1983, Noss 1983). Fencerows may therefore contain only a subset of avian species adapted to small fragments, edge and agriculture.

Forest fragmentation results in habitat loss and increasing isolation of the remaining forest fragments. Fragment size determines the potential size of populations

(Saunders *et al.* 1991) and woodlot area has been shown to be positively associated with avian species richness and composition (Galli *et al.* 1976, Whitcomb *et al.* 1981, Ambuel & Temple 1983, Opdam *et al.* 1985, Askins *et al.* 1987, Robbins *et al.* 1989). Isolated fencerows may have lower species richness due to decreased immigration rates from nearby fragments (Merriam & Wegner 1992). The amount of wooded habitat in the surrounding area may also influence immigration; areas with more wooded habitat provide a larger source of immigrants to help sustain populations in individual fragments (Askins *et al.* 1987, Robinson *et al.* 1995).

Many species in small habitat fragments in an agricultural landscape use the resources from surrounding fields and these fields may constitute a productive part of the home range, resulting in the habitat used being larger than the actual wooded habitat measured (Blake & Karr 1984, Loman & von Schantz 1991). The habitat surrounding a fencerow may therefore be important in determining use, with crop type influencing species richness and abundance (Shalaway 1985, Green *et al.* 1994, Parish *et al.* 1994).

The aim of this study was to determine the relationship between fencerow and landscape characteristics and avian species composition in fencerows in an agricultural landscape in central Alberta. The fencerow characteristics examined were area, vegetation complexity and isolation, and the landscape characteristics were landscape type (agricultural or forested), landscape context (amount of wooded habitat surrounding the fencerow measured at two scales) and adjacent land use. To determine if the species found in fencerows were representative of those found in woodlots, the results were compared to another study on birds in woodlots in the same area (Hannon unpubl.).

3.1 METHODS

3.1.1 Study area

Thirty fencerows in a predominantly agricultural region surrounding the Meanook Biological Research Station (54° 37' N, 113° 20' W), near Athabasca, Alberta, Canada, were studied during the summers of 1993 and 1994. Fencerows were approximately 800m long, ranged in width from 5-17m and were dominated by trembling aspen (*Populus tremuloides*) and balsam poplar (*P. balsamifera*). Understory vegetation was dominated by rose (*Rosa spp.*), raspberry (*Rubus idaeus*), red-osier dogwood (*Cornus stulonifera*) and saskatoon (*Amelanchier alnifolia*). I chose two landscape types based on the percentage forest cover in a 125km^2 area: 1) agricultural landscape with 20% forest cover, and 2) forested landscape with 48% forest cover. Within each landscape type, 6 fencerows were connected to a woodlot and 9 fencerows were isolated (no connection to a woodlot, range 80-829m from closest woodlot).

3.1.2 Avian census

A 500m long section in the centre of each fencerow was surveyed. Surveys were conducted during the breeding season from late May to the end of June between 5:00-10:00am. Birds were surveyed by slowly walking down one edge and stopping for one minute every 50m. The locations of all bird species seen or heard within the fencerow were recorded, and due to the narrow width of fencerows, it was assumed that all species were seen or heard. Fencerow, observers, and starting location and time were randomized to minimize biases. Surveys were conducted 4 times in 1993. At least one singing male had to be recorded over the 4 dawn censuses for the species to be recorded as present in

the fencerow (# singing males) and the total number of species detected (visually or aurally) was also recorded for each fencerow (# species detected). In 1994, the number of surveys was increased to 8 and a minimum of 2 registrations separated by 10 days was the criteria used to delineate a territory (Bibby *et al.* 1992). The total number of species holding a territory in each fencerow (# of species with a territory), territory density, and the total number of species detected in each fencerow, including territory holders and casual sightings (# species detected) were recorded from the spot maps. Although an important point of concern is whether species are establishing territories and breeding successfully in fencerows, records of transients may provide additional information regarding the use of fencerows for additional purposes e.g. inclusion of more than one fragment within a territory, foraging from nearby woodlots and for movement. Analyses were therefore also performed on variables including transients (# species detected in 1993 and 1994).

3.1.3 Vegetation

Vegetation was sampled using the point-quarter method (Cottam & Curtis 1956) every 20m along a medial transect within the bird census section of each fencerow. Shrub and tree density and species composition were determined using the 'Quarter Program' (Krebs 1989). Canopy cover and average tree canopy height were recorded at every 20m station using a spherical densiometer and clinometer respectively. Percent ground cover, leaf litter depth and percent cover, average herbaceous height and percent cover were recorded within a 1m x 20cm quadrat.

3.1.4 Fencerow and landscape characteristics

Landscape and fencerow characteristics were recorded from 1:30,000 aerial photographs taken in 1995 using a Summagraphics SummaSketch digitiser (Model MM1201) and Sigma-Scan (1988). Information was referenced against 1:25,000 photographs taken in 1992 and verified in the field to provide current details. The following fencerow and landscape characteristics were recorded from the photographs:

1) Area (ha) of the fencerow (AREA)

2) Area (ha) of wooded habitat in a rectangular area $2km^2$ around the fencerow (AREA2)

3) Area (ha) of wooded habitat in a rectangular area $15km^2$ around the fencerow (AREA15)

Adjacent crop types were recorded. Hay, alfalfa and pasture were combined into a hay category, and oats and barley combined into a grain category.

3.1.5 Fencerows vs woodlots

In 1991 and 1992 bird species were recorded during point counts in 16 woodlots (range 0.7-140ha) in the area surrounding Meanook Biological Research Station (Hannon unpubl.). These data were used to represent the local species pool and a comparison made with the species found in fencerows. The migratory status and habitat types of the species found in fencerows and woodlots were compared. Migratory status was defined as : (1) resident - remain in the area throughout the year, (2) short-distance - winter south of the area but north of the tropics, and (3) neotropical - winter in the tropical and subtropical areas of Central and South America (Freemark & Collins 1989). Habitat use classifications followed Freemark & Collins (1989): (1) interior - nest within the interior of the forest and are rarely found near edges, (2) interior-edge - have territories located

within the forest but also use the edge, (3) edge - use fragment edge, fields, or openings within a fragment during the breeding season. I added an additional category of open species which usually nest in grassy areas and agricultural fields and are not usually associated with wooded habitat but used fencerows as part of their territory, e.g. for song posts and foraging, and often nested in the grassy strip between the fencerow and crop.

3.1.6 Statistical analyses

3.1.6.1 Fencerow vs woodlot

G-tests were performed to determine if woodlots and fencerows differed in the migratory status and habitat type for the number of recorded breeding bird species.

3.1.6.2 Landscape type and isolation

To determine if the main effects of the study design affected avian abundance and richness, comparisons between landscape type (agricultural/forested) and isolation (isolated/connected) were performed for the recorded avian variables (section 3.1.2) using two-way Kruskal-Wallis analyses. To ensure that the results were not an artifact of differing vegetation types, the vegetation variables were initially compared among the 4 groups and the conclusion drawn that the vegetation was relatively homogeneous within fencerows between the main effects groups (see Chapter 2).

3.1.6.3 Fencerow and landscape characteristics

3.1.6.3.1 Species richness and abundance

The relationships between species richness/abundance, and the fencerow and landscape characteristics (Table 3.1) were examined using stepwise multiple regression analyses (SYSTAT 1992). Stepwise multiple regression is a data exploration technique

used to determine a subset of predictor variables which combine parsimony with adequacy of prediction (Sokal & Rohlf 1981). Dependent variables were transformed when necessary to approximate normal distributions. For each dependent variable, in all circumstances possible, linearity and homogeneity of variance with each independent variable was achieved. Area variables were square root transformed. Variables which are correlated can result in the partial regression coefficients associated with them not reflecting the true dependence of the dependent variable on the independent variable, giving spurious results (Zar 1984). Area of wooded habitat within $2km^2$ (AREA2) and $15km^2$ (AREA15) were correlated, therefore, two different regressions were run, including either AREA2 or AREA15, and the model with the best parameter estimates was chosen. There were significant correlations between herbaceous cover and canopy cover (negative) and herbaceous height (positive), and between rock cover and leaf cover (negative). Herbaceous cover and rock cover were omitted from further analyses. The two crop categories (hay and grain) encompassed the crops adjacent to the fencerows and were inversely related, therefore they were represented in analyses as one category (HAY) where the percent of hay was coded as dummy variables. A stepwise regression analysis was initially performed for a dependent variable including only the fencerow characteristics. The residuals from this analysis were saved, and a further stepwise regression analysis performed on these residuals using the landscape characteristics. This determines initially the fencerow characteristics accounting for variation in avian variables, and then shows what additional effects are explained by the landscape variables. Residual plots were examined for violations of assumptions and adequacy of the fit of the model.

3.1.6.3.2 Individual species

Ordination analysis was then used to relate patterns in the avian community composition to the fencerow and landscape characteristics (environmental variables). Ordination techniques assume either a linear or unimodel species response to environmental gradients. In order to select the appropriate technique, detrended correspondence analysis (DCA) was used to calculate the length of the environmental gradients (ter Braak 1990). Short gradient lengths (<2 standard deviations from the mean) indicate that the species responses are typically linear (ter Braak 1987). The length of the gradient of the first axis was 1.59 indicating that a linear model would be the most appropriate to detect patterns in the data. Redundancy analysis (RDA), a linear direct gradient technique, was used to explore the effect of the fencerow and landscape characteristics (Table 3.1) on species composition. The number of territories for each species per fencerow (recorded in 1994) was used to be representative of the species composition. The 16 species which established territories were included in the analysis (Table 3.2). The analysis was performed using CANOCO (version 3.12; ter Braak 1990). Fencerow and landscape characteristics that contributed the most to the explanation of species abundance were chosen by the forward selection procedure ($p<0.1$). The forward selection procedure aims to select a minimum number of environmental variables that explain the species data as well as the full set (ter Braak 1990). Environmental variables that were highly multicollinear (variance inflation factor >20; ter Braak 1990) were excluded from the analysis.

3.2 RESULTS

3.2.1 Fencerow vs woodlot

A total of 39 species were detected in fencerows (Appendix 1), sixteen of which were identified as having established territories in 1994 (Table 3.2). The most abundant species with territories were Song Sparrows, Clay-coloured Sparrows, Red-eyed Vireos and Black-capped Chickadees (Appendix 1). Anecdotal evidence indicates that some species were breeding in the fencerows (nests or fledglings found) (Appendix 1). Fifty-nine species were recorded in woodlots during 1991 and 1992 (Appendix 2).

There were no differences in the number of species of the differing migratory classifications (resident, short-distance, neotropical) between woodlots and fencerows (Table 3.3, G=0.86, d.f.=2, p=0.65). There were no differences in the numbers of species within the different habitat classifications (interior and interior/edge, edge and edge/open, and open) between woodlots and (Table 3.3, G=5.48, d.f.=2, p=0.06). Examination of Table 3.3 indicates that a difference between fencerows and woodlots for interior and interior/edge classifications might be expected. Because the result was marginally significant I analyzed further. There was a significant difference in interior and interior/edge combined when compared to open, edge, and edge/open combined, with fewer interior and interior/edge species in the fencerows (G=5.48, d.f.=1, p=0.02). There were no differences between edge species and interior, interior/edge and open species combined (G=2.34, d.f.=1, p=0.13) or between open species and the interior, interior/edge and edge species combined (G=1.12, d.f.=1, p=0.29).

3.2.2 Landscape type and isolation

In 1993 there were no significant effects of landscape type (Table 3.4), isolation (Table 3.5) or interaction on the number of singing males or the number of species detected. In 1994 there was no significant effect of landscape type (Table 3.4) or isolation (Table 3.5) on the number of territories or the number of species detected. However, there was a marginally significant isolation effect on territory density with densities being higher in the isolated fencerows (Table 3.5, H=3.88, d.f.=1, $H_{critical0.05,2}$=3.841). There were no significant interactions between landscape type and isolation for territory density or for the number of territories, however, there was a significant interaction between landscape type and isolation for the number of species detected (Figure 3.1) with more species in the connected fencerows in the forested landscape than in the agricultural landscape and the opposite occurred in isolated fencerows.

3.2.3 Fencerow and landscape characteristics

3.2.3.1 Species richness and abundance

The stepwise regression analyses indicates that fencerow characteristics are the only determinants of species richness and abundance (Table 3.6), however, these predictor variables only explain a low amount of the variation within the data.

3.2.3.2 Individual species

The ordination biplot (Figure 3.2) represents the relationship between avian territory abundance and the fencerow and landscape characteristics found in Table 3.1. The eigenvalues of the first and second axes were 0.13 and 0.07, respectively. The axes explained 20.4% of the total variation in the species data. The fraction of this variance accounted for by the ordination is 90.5% i.e. the first 2 axes explained 91% of the variance

in the species-environment relationship. The species-environment correlations were 0.69 and 0.65 for the first and second axes, respectively, indicating a high relation between the species and environment for each axis. Only three variables were retained after the forward selection procedure with AREA15 contributing most strongly to axis 1 and TREE DIVERSITY and SHRUB DENSITY to axis 2 (Table 3.7). Species in the centre of the species-environment biplot (Figure 3.2) are not well represented and thus any inferences may be imprecise (ter Braak. 1987). American Robins, Black-capped Chickadees and White-throated Sparrows were found mainly in fencerows with high shrub density and surrounded by a high percent of wooded habitat within $15km^2$. Mourning Warblers were associated with a high amount of wooded habitat within $15km^2$. Vesper Sparrows were found mainly in fencerows with high tree species diversity and were negatively affected by wooded habitat within $15km^2$. Red-eyed vireos and Clay-coloured Sparrows were found in fencerows with high tree diversity and shrub density, however, they differed in their association with wooded habitat within $15km^2$ with a positive and a negative assocaition for Red-eyed Vireos and Clay-coloured Sparrows respectively. Song Sparrow and Northern Flicker territory abundance were negatively related to the area of wooded habitat within $15km^2$. The Eastern Kingbird was found in fencerows with low shrub density.

3.3 DISCUSSION

3.3.1 Community structure

Establishment of territories and anecdotal evidence demonstrates that several species use these habitats as breeding habitat. Only a subset of the local forest species pool was recorded in fencerows with 16 species establishing territories. The species

which established territories were predominantly edge or open species. There were proportionately fewer interior/edge species in fencerows than in woodlots and no interior species established territories.

Edge species, farmland species and generalists are common in small fragments (Ambuel & Temple 1983, Lynch & Whigham 1984). Askins *et al.* (1987) postulated that the small size of fragments may be contributing to population declines of forest-interior species due to a number of factors: increased edge with corresponding negative edge effects, fewer microhabitats available, smaller than minimum territory size for a particular species, and increased probability of extinction through stochasticity. Fencerows are entirely edge habitat (Noss 1983, Bennett 1990) with insufficient habitat available to support interior species. The increased competition, predation and parastism associated with edge may exclude some species (Ambuel & Temple 1983).

3.3.2 Landscape type and isolation

Birds found in individual patches may be influenced by both the amount of wooded habitat in the area (Askins *et al.* 1987, van Dorp & Opdam 1987, Freemark & Collins 1989) and the spatial arrangment of fragments (Pearson 1993). Area-sensitive species increased with forest cover and the number of interior species increased the most rapidly with fragment size in areas with greater forest cover (Freemark & Collins 1989). However, in this study there was no effect of the amount of forest cover on species richness or abundance. This suggests that either the landscape is not important for these bird species or that other landscape-level characteristics play a role in affecting species abundance e.g. spatial configuration.

Isolation was not an important factor in this study. Wegner & Merriam (1979) reported that in an agricultural landscape the majority of bird movements were between fencerows and woodlots with few species crossing open fields. Other studies have also suggested that linear strips may be movement corridors for bird species (e.g. Haas 1995, Machtans *et al.* in press). The fencerows in this landscape were interconnected, forming a network, and species may be using fencerows as both habitat and conduits. The interaction for the number of species detected showed more species in connected fencerows in the forested landscape than in the agricultural landscape, and the opposite in the isolated fencerows. This indicates a tendancy for species to use connected fencerows as an extension of woodlots in the forested landscape, but may also indicate potential use for movement.

Possible explanations for there being no effect of landscape type or isolation in my study include: (1) insufficient variation in the landscape features e.g birds are highly mobile and the isolation distances in this area may not have inhibited movements, (2) species are using surrounding habitats to provide resources and are not confined to only wooded habitat, (3) the species found in this landscape are adapted to this level of habitat heterogeneity and fragmentation.

3.3.3 Fencerow and landscape characteristics

3.3.3.1 Species richness and abundance

Within the range of fencerow and landscape characteristics measured in this study, avian richness and abundance, and abundance of individual species' territories are determined primarily by vegetation characteristics, and are relatively uninfluenced by

landscape scale components. McGarigal & McComb (1995) found that landscape configuration explained only a moderate amount of the variation in avian abundances within habitats. Pearson (1993) found that the landscape diversity determined the species pool of wintering birds while local vegetation structure determined the suitability of a habitat for each species.

Vegetation characteristics were the primary predictors of avian richness and abundance in fencerows. Habitat complexity and heterogeneity have been shown to be important determinants for many bird species (e.g. Lynch & Whigham 1984, Freemark & Merriam 1986, Blake & Karr 1987). In fencerows, shrub density and tree diversity were important characteristics. This might be expected as 66% of the species recorded were shrub, tree or cavity nesters. Herbaceous height and cover were predictors of species richness in 1993, providing cover and protection for the ground nesting species. It is unclear why differing characteristics were predictors for the number of species detected in each year.

Fragment size determines the potential population size of different species (Saunders *et al.* 1991) with species richness and density increasing with area (e.g. Whitcomb *et al.* 1981, Blake & Karr 1984, Freemark & Merriam 1986, Askins *et al.* 1987, Hannon 1992,). Territory density was higher in smaller fencerows. This result is supported by a few studies of bird density in woodlots (e.g. Helle 1985, Lynch & Whigham 1984, Loman & von Schantz 1991). Due to the linear nature of the habitat and high visibility of adjacent habitats, the tendancy to overestimate territories in smaller areas is probably not a concern. In agricultural areas, the crops may be an integral part of an

individual's territory resulting in the habitat area being more extensive than the actual fencerow (Loman & von Schantz 1991). In addition, leaf litter depth was an important determinant of territory density. Of those species which established territories, 31% and 63% were primarily ground nesters and ground foragers respectively. Species which forage and nest on the ground require a layer of litter on the soil surface (Erksine 1977).

3.3.3.2 Individual species

Local vegetation characteristics also influenced individual species territory abundance. Shrub density and tree diversity were primary characteristics influencing individual species' territory abundances in fencerows, providing nesting sites, protection and food. The amount of wooded habitat within $15km^2$ was also an important predictor for some species, especially Song and Clay-coloured Sparrows. Both species can nest in open areas and were higher in fencerows with increasing agricultural land in the surroundings. However, Song Sparrows rarely forage in fields (Yahner 1983) and fencerows provide foraging habitat. American Robins nest in trees and shrubs and were associated with fencerows with high shrub density and tree diversity. Optimal habitat for Red-eyed Vireos is characterized by dense foliage, with high foliage height and openings in the canopy (Ambuel & Temple 1983) and they can use both forest interior and edge (Whitcomb *et al.* 1981). Fencerows with high shrub density and tree diversity provided suitable habitat for establishing territories. Mourning Warblers breed primarily in dense shrubs in open deciduous woods, nesting at the base of a shrub either on or near the ground (Ehrlich *et al.* 1988). My results suggest that this is important as Mourning Warblers established territories in fencerows with high shrub density. Mourning Warblers,

White-throated Sparrows, American Robins and Black-capped Chickadees territories were established in fencerows with a greater amount of wooded habitat in the surrounding area, suggesting that dispersal of these species from other wooded habitat may be important (Askins *et al.* 1987).

3.3.4 Fencerows as habitat

The distribution of birds in fragmented landscapes is determined primarily by the presence of suitable habitat (Opdam *et al.* 1985). Fencerows provide habitat for some species adapted to edge but are unable to support many of the species in the local species pool. Gates & Gysel (1978) showed that field-forest edges may have high species richness and abundance but these species were subjected to high rates of cowbird parasitism and predation. Hannon (unpubl. data) found 100% predation on artificial nests in fencerows in this area. Further research is required to determine whether fencerows are indeed "ecological traps" (Gates & Gysel 1978). Species which do not use fencerows as breeding habitat may still benefit from their presence in the landscape if they are used as movement corridors, to relieve isolation between suitable habitat fragments. In landscapes with few woodlots, fencerows may play an important role in the dispersal, and consequent survival, of many species (Forman & Baudry 1984). But, there may also be costs associated with fencerow corridors through the enhancement of disease transmission, spread of fire, predators and opportunistic species (Simberloff & Cox 1987).

Data on the reproductive success of bird species and the use of fencerows as corridors are required before any conclusions can be made regarding the benefits of fencerows to bird species in agricultural landscapes. Fencerows are a valuable addition to

the landscape but cannot be regarded as an alternative to woodlots in the provision of habitat for many bird species.

Table 3.1: Fencerow and landscape characteristics (environmental variables) included in the redundancy analysis (RDA) of avian species composition.

Landscape type: Agricultural/Forested (categorical variables)
Connected/Isolated (categorical variables)
Area (ha) of fencerow
Area (ha) of wooded habitat within $2km^2$
Area (ha) of wooded habitat within $15km^2$
Hay in adjacent fields (categorical variable[1]: 1=0%, 2=50%, 3=100%)
Canopy cover (%) - mean of the measurements taken at every 20m station
Canopy height (m)- mean of the measurements taken at every 20m station
Herbaceous cover (%) - mean of the measurements taken at every 20m station
Herbaceous height (cm) - mean of the measurements taken at every 20m station
Rock cover (%) - mean of the measurements taken at every 20m station
Leaf cover (%) - mean of the measurements taken at every 20m station
Leaf depth (cm) - mean of the measurements taken at every 20m station
Shrub density - mean of the measurements taken at every 20m station
Tree density - mean of the measurements taken at every 20m station
Shrub diversity - mean of the measurements taken at every 20m station
Tree diversity - mean of the measurements taken at every 20m station

[1] The crop type in fields adjacent to each fencerow were classified as 0% if neither field contained hay, 50% if one field contained hay, and 100% if there was hay in both fields.

Table 3.2: Species with established territories and included in the redundancy analysis (RDA) of avian composition (1994).

Species	*Scientific name*
Song Sparrow (SOSP)	*Melospiza melodia*
Clay-coloured Sparrow (CCSP)	*Spizella pallida*
White-throated Sparrow (WTSP)	*Zonotrichia albicollis*
Red-eyed Vireo (REVI)	*Vireo olivaceus*
Black-capped Chickadee (BCCH)	*Parus atricapillus*
Savannah Sparrow (SVSP)	*Passerculus sandwichensis*
American Robin (AMRO)	*Turdus migratorius*
American Crow (AMCW)	*Corvus brachyrhynchos*
Vesper Sparrow (VESP)	*Pooecetes gramineus*
House Wren (HOWR)	*Troglodytes aedon*
Black-billed Magpie (BBMA)	*Pica pica*
European Starling (EUST)	*Sturnus vulgaris*
Eastern Kingbird (EAKI)	*Tyrannus tyrannus*
Northern Flicker (NOFL)	*Colaptes auratus*
Yellow Warbler (YEWA)	*Dendroica petechia*
Mourning Warbler (MOWA)	*Oporornis philadelphia*

Table 3.3: Species recorded in woodlots and fencerows classified according to their migratory status (RES, resident; SD, short-distance; NTM, neotropical migrant) and habitat use (I, interior; I/E, interior-edge; E, edge; E/O, edge and open; O,open).

	Migratory status				**Habitat use**			
	RES	SD	NTM	*Total N*	I and I/E	E and E/O	O	*Total N*
Woodlot (%)	19	37	44	*59*	51	41	8	*59*
Fencerow (%)	25	44	31	*16*	19	62	19	*16*

Table 3.4: Avian variables (mean±S.E.) for the landscape types. Comparisons between the landscape types (agricultural/forested) were made using Kruskal-Wallis analyses.

	Landscape type			
Avian variable	Agriculture (n=15)	Forested (n=15)	H	p
1993				
# singing males	3.33±0.27	2.9±0.15	≤0.72	>0.5
# species detected	7.07±0.36	6.4±0.42	≤0.59	>0.25
1994				
territory density	10.19±1.20	8.69±1.08	≤0.56	>0.25
# territories	3.53±0.41	4.4±0.52	≤1.12	>0.25
# species detected	8.07±0.75	8.8±0.63	≤0.65	>0.25

Table 3.5: Avian variables (mean±S.E.) for the connected and isolated fencerows. Comparisons between the fencerow types (connected/isolated) were made using Kruskal-Wallis analyses.

	Fencerow type			
Avian variable	Connected (n=15)	Isolated (n=15)	H	p
1993				
# singing males	3.17±0.24	3.11±0.21	≤0.09	>0.9
# species detected	7.33±0.38	6.33±0.37	≤1.84	>0.1
1994				
territory density	7.54±1.06	10.71±1.06	≤3.88	<0.05
# territories	4.08±0.68	3.89±0.34	≤0.02	>0.9
# species detected	8.5±0.96	8.39±0.53	≤0.002	>0.5

Table 3.6: Stepwise multiple regression analyses of avian species richness and abundance (1993/1994) in fencerows. Subsets of predictor variables for each avian variable and the variance explained are shown for the fencerow characteristics. There were no landscape predictors.

Avian variable	Fencerow characteristic	% variance explained
1993		
# males	(1) Shrub density (+)	9
# species detected	(1) Herbaceous height (+)	15
1994		
territory density	(1) Area (-) (2) Leaf depth (+)	38
# species with territories	(1) Tree diversity (+)	17
# species detected	(1) Tree diversity (+)	20

Table 3.7: Summary statistics for redundancy analysis (RDA) showing the contribution of the fencerow and landscape characteristics to axes 1 and 2 of the avian territory abundance in fencerows.

	Canonical coefficient		Correlation coefficient	
Variable	Axis 1	Axis 2	Axis 1	Axis 2
AREA15	0.90	-0.26	0.65	-0.19
TREE DIVERSITY	0.10	0.72	-0.09	0.48
SHRUB DENSITY	0.31	0.68	0.35	0.38

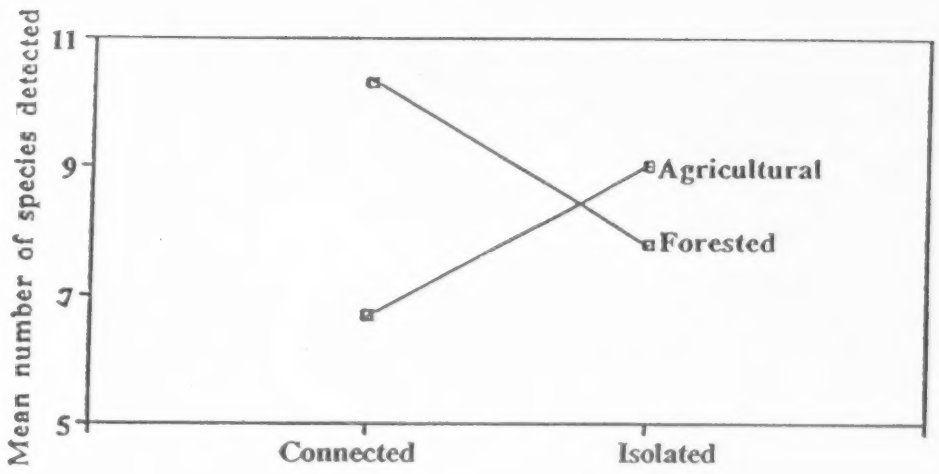

Figure 3.1: Interaction plot between landscape type (agricultural/forested) and fencerow type (connected/isolated) for the mean number of species detected in 1994 (Kruskal-Wallis H=5.30, p=0.05).

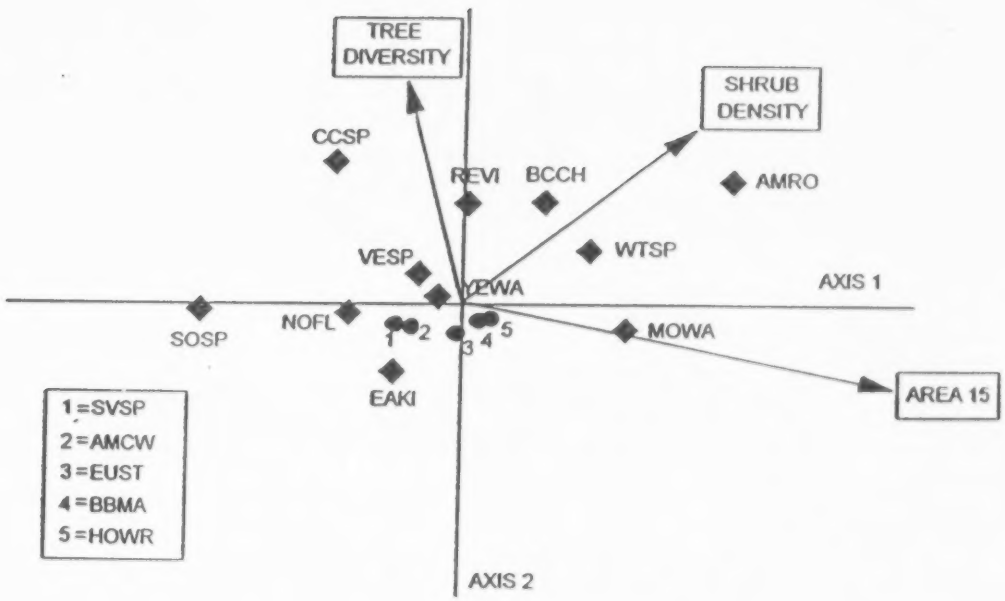

Figure 3.2: Redundancy analysis ordination biplot of avian territory abundance and the fencerow and landscape characteristics. Avian species codes: SOSP, Song Sparrow; CCSP, Clay-coloured Sparrow; WTSP, White-throated Sparrow; REVI, Red-eyed Vireo; BCCH, Black-capped Chickadee; SVSP, Savannah Sparrow; AMRO, American Robin; AMCW, American Crow; VESP, Vesper Sparrow; HOWR, House Wren; BBMA, Black-billed Magpie; EUST, European Starling; EAKI, Eastern Kingbird; NOFL, Northern Flicker; YEWA, Yellow Warbler; MOWA, Mourning Warbler.

3.4 LITERATURE CITED

Ambuel, B., and S.A. Temple. 1983. Area-dependent changes in the bird communities and vegetation of southern Wisconsin forests. Ecology **64(5)**:1057-1068.

Askins, R.A., M.J. Philbrick, and D.S. Sugeno. 1987. Relationship between the regional abundance of forest and the composition of forest bird communities. Biological Conservation **39**: 129-152.

Bennett, A.F. 1990. Habitat corridors: their role in wildlife management and conservation. Arthur Rylah Institute for Environmental Research. Department of Conservation and Environment, Victoria, Australia.

Bibby, C.J., N.D. Burgess, and D.A. Hill. 1992. Bird census techniques. Academic Press, Toronto. 257pp.

Blake, J.G., and Karr, J.R. 1984. Species composition of bird communities and the conservation benefit of large versus small forests. Biological Conservation **30**: 173-187

Blake, J.G., and Karr, J.R. 1987. Breeding birds of isolated woodlots: area and habitat relationships. Ecology **68(6)**: 1724-1734.

Brittingham, M.C., and Temple, S.A. 1983. Have cowbirds caused forest songbirds to decline? BioScience **33**: 31-35.

Cottam, G., and J.T. Curtis. 1956. The use of distance methods in phytosociological sampling. Ecology **37**: 451-460.

Ehrlich, P.R., D.S. Dobkin, and D. Wheye. 1988. The birder's handbook: a field guide to the natural history of North American birds. Simon and Schuster Inc., New York, USA.

Erskine, A.J. 1977. Birds in boreal Canada: communities, densities and adaptations. Canadian Wildlife Service Report Series Number 41.

Forman, R.T.T., and J. Baudry. 1984. Hedgerows and hedgerow networks in landscape ecology. Environmental Management **8(6)**: 495-510.

Freemark, K.E., and B. Collins. 1989. Landscape ecology of breeding birds of birds breeding in temperate forest fragments. Pages 443-454 *in* J.M. Hagan III and D.W. Johnston, editors. Ecology and conservation of neotropical migrant landbirds. Smithsonian Institution Press, Washington, D.C., USA.

Freemark, K.E., and H.G. Merriam. 1986. Importance of area and habitat heterogeneity to bird assemblages in temperate forest fragments. Biological Conservation **36**: 115-141.

Galli, A.E., C.F. Leck, and R.T.T. Forman. 1976. Avain distribution patterns in forest islands of different sizes in central New Jersey. The Auk **93**: 356-364.

Gates, J.E., and L.W. Gysel. 1978. Avian nest dispersion and fledgling success in field-forest ecotones. Ecology **59(5)**: 871-883.

Green, R.E., P.E. Osborne, and E.J. Sears. 1994. The distribution of passerine birds in hedgerows during the breeding season in relation to characteristics of the hedgerow and adjacent farmland. Journal of Applied Ecology **31**: 677-692.

Haas, C.A. 1995. Dispersal and use of corridors by birds in wooded patches on an agricultural landscape. Conservation Biology **9**: 845-854.

Hannon, S.J. 1992. Nest predation and forest bird communities in fragmented aspen forests in Alberta. Pages 127-136 *in* D.H. Kuhnke, editor. Birds in the boreal forest: proceedings of a workshop held March 10-12, 1992 in Prince Albert, Saskatchewan. Northern Forestry Centre, Forestry Canada, Northwest Region, Edmonton, Alberta.

Henderson, M.T., G. Merriam, and J. Wegner. 1985. Patchy environments and species survival: chipmunks in a agricultural mosaic. Biological Conservation **31**: 95-105.

Helle, P. 1985. Effects of forest fragmentation on bird densities in northern boreal forests. Ornis Fennica **62**: 35-41.

Krebs, C.J. 1989. Ecological Methodology. Harper Collins Publishers, New York. 654pp.

Loman, J., and T. von Schantz. 1991. Birds in farmland - more species in small than in large habitat island. Conservation Biology **5(2)**: 176-188.

Lynch, J.F., and D.F. Whigham. 1984. Effects of forest fragmentation on breeding bird communities in Maryland, USA. Biological Conservation **28**: 287-324.

Machtans, C.S., M.A. Villard, and S.J. Hannon. In press. Use of riparian buffer strips as movement corridors by forest birds. Conservation Biology.

McGarigal, K., and W.C. McComb. 1995. Relationships between landscape structure and breeding birds in the Oregon Coast Range. Ecological Monographs **65(3)**: 235-260.

Merriam, G., and J. Wegner. 1992. Local extinctions, habitat fragmentation, and ecotones. Pages 150-169 *in* A.J. Hansen and F. di Castri, editors. Landscape boundaries: consequences for biotic diversity and ecological flows. Springer-Verlag, New York, Inc.

Noss, R.F. 1983. A regional landscape approach to maintain diversity. BioScience **33(11)**: 700-706.

Opdam, P., G. Rijsdijk, and F. Hustings. 1985. Bird communities in small woods in an agricultural landscape: effects of area and isolation. Biological Conservation **34**: 333-352.

Parish, T., K.H. Lakhani, and T.H.Sparks. 1994. Modelling the relationship between bird population variables and hedgerow and other field margin attributes. I. Species richness of winter, summer and breeding birds. Journal of Applied Ecology **31**: 764-775.

Pearson, S.M. 1993. The spatial extent and relative influence of landscape-level factors on wintering bird populations. Landscape Ecology **8(1)**: 3-18.

Robbins, C.S., J.R. Sauer, R.S. Greenberg, and S. Droege. 1989. Population declines in North American birds that migrate to the neotropics. Proceedings of the National Academy of Science USA **86**: 7658-7662.

Robinson, S.K., F.R. Thompson III, T.M. Donovan, D.R. Whitehead, and J. Faaborg. 1995. Regional forest fragmentation and the nesting success of migratory birds. Science **267**: 1987-1990.

Saunders, D.A., R.J. Hobbs, and C.R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology **5(1)**: 18-32.

Shalaway, S.D. 1985. Fencerow management for nesting birds in Michigan. Wildlife Society Bulletin **13(3)**: 302-306.

Sigma-Scan V3.90. 1988. The Scientific Measurement Program. Jandel Scientific.

Simberloff, D., and J. Cox. 1987. Consequences and costs of conservation corridors. Conservation Biology **1(1)**: 63-71.

Sokal, R.R., and F.J. Rohlf. 1981. Biometry. 2nd Edition. W.H. Freeman and Company, New York. 859pp.

SYSTAT for windows: statistics, version 5 edition. Evanston IL:SYSTAT, Inc., 1992.

ter Braak, C.F.R. 1987. Ordination. Pages 91-15 *in* R.H.G. Jongman, C.F.R. ter Braak, and O.F.R. van Tongeran, editors. Data analysis in community and landscape ecology. Pudoc Wageningen, Netherlands.

ter Braak, C.F.R. 1990. Update notes: CANOCO version 3.10. Agricultural Mathematics Group, 6700 AC, Wageningen, Netherlands.

van Dorp, D., and P.F.M. Opdam. 1987. Effects of patch size, isolation and regional abundance on forest bird communities. Landscape Ecology **1(1)**: 59-73

Wegner, J.F., and G. Merriam. 1979. Movements by birds and small mammals between a wood and adjoining farmland habitats. Journal of Applied Ecology **16**: 349-357.

Whitcomb, R.F., C.S. Robbins, J.F. Lynch, B.L. Whitcomb, M.K. Klimliewcz, and D. Bystrak. 1981. Effects of forest fragmentation on avifauna of the eastern deciduous forest. Pages 125-205 *in* R.C. Burgess and D.M. Sharpe, editors. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York.

Yahner, R.H. 1982. Avian nest densities and nest-site selection in farmstead shelterbelts. Wilson Bulletin **94(2)**: 156-175.

Yahner, R.H. 1983. Seasonal dynamics, habitat relationships, and management of avifauna in farmstead shelterbelts. Journal of Wildlife Management **47(1)**: 85-104.

Zar, J.H. 1984. Biostatistical analysis Second edition.Prentice Hall, Englewood Cliffs, New Jersey. 718pp.

CHAPTER 4

Thesis Discussion

4.0 DISCUSSION

Habitat fragmentation results in loss of the original habitat, reduction in fragment size, and increased isolation of fragments, causing a decline in species diversity within the original habitat (Andren 1994). The main focus of this research was to investigate two of these factors by determining how the proportion of remaining wooded habitat and isolation affect bird and small mammal species found in fencerows. This landscape has been fragmented for approximately 100 years (Stone 1969) and it is probable that many species susceptible to fragmentation have already become extirpated and that the remaining species are adapted to this level of habitat alteration.

Landscape characteristics and fencerow isolation did not affect the abundance and richness of bird species found in fencerows, only local characteristics (fencerow vegetation) influenced avian variables. In contrast, small mammal species were influenced primarily by landscape type, however, species responded differently. *P. maniculatus* abundance was higher in the agricultural landscape and relatively unaffected by isolation. *C. gapperi* is a woodland specialist (Yahner 1983 & 1992) and this is reflected in the results with higher abundances in the forested landscape and in connected fencerows during the year of relatively higher densities. This indicates that fencerows may be marginal habitat supporting the surplus individuals from the woodlots.

These differences may result from the differing scales of use of the landscape by species e.g. home range size and dispersal distances. Birds are highly mobile organisms and their perception of this level of fragmentation may differ from that of small mammals. Birds may not perceive my two classifications of landscape type as substantially different,

instead using habitat cues in their territory selection. The threshold at which organisms are affected by isolation and landscape context may be lower for bird species due to their mobility. For example, Haas (1995) found that bird species of open landscapes used corridors for movements but that these corridors were not essential for dispersal. Small mammal species movements are more likely inhibited by agricultural fields than birds, resulting in the landscape type and configuration exerting a greater influence on small mammals.

4.1 Fencerows as habitat

Linear strips are considered to be important to wildlife (e.g. Yahner 1982 & 1983, Forman & Baudry 1984, Henderson *et al.* 1985). In an area of extensive agriculture with few woodlots, fencerows provide habitat for many species. Due to their small size they are essentially all edge (Noss 1983) and may be functioning as "ecological traps" for some species (Gates & Gysel 1978). Fencerows do not support all species from the local forest pool (e.g. species that require large tracts of habitat) and the species which are abundant in fencerows are generally widespread, common species. Fencerows alone will not maintain the local species pool, other larger wooded fragments are necessary to prevent species loss. Both grazing and removal of woodlots should be discouraged if current species are to be retained in this landscape

4.2 Future research

Future research should investigate the crop and wooded fragment mosaic as a whole system and determine the contribution of each habitat type to birds and small mammals. Species may require resources from more that one patch type or include

several habitat types within their home range. Effective conservation in an agricultural landscape will require a mosaic of different habitat fragments that together will support substantial regional populations (Henderson *et al.* 1985, Bennett 1987).

Fencerows may be habitat sinks which are supported by source populations from nearby woodlots. Data are needed on the reproductive success of species and the effects of predation and parasitism. If fencerows are habitat sinks they may still have potential as conservation tools if they are conduits for species, thus reducing isolation between habitats. Dispersal is a process central to the survival of species in fragmented landscapes, and fencerows may benefit many species by reducing the effective isolation distance. Research is needed to establish if fields inhibit dispersal and if fencerows relieve this barrier by facilitating movements.

This study is a small snapshot, both spatially and temporally, of bird and small mammal abundance in this landscape. Longer term and more spatially extensive studies are needed to determine the effects of fragmentation on birds and small mammals in this area.

4.3 Management implications

This study indicates that continued destruction of woodlots through logging and cattle grazing will result in loss of species from the present local species pool as fencerows only contain a subset of this species pool. Limiting grazing to a few woodlots, increasing use of pastures for grazing and allowing regeneration of some sections of land will increase the likelihood of retaining this species pool. Fencerows were used as habitat by many species, therefore retention of these linear habitats is advocated. Increasing the area

of habitat available, by widening the fencerows, would also be beneficial. Conservation of isolated fencerows will likely not sustain many species over the long term, therefore retaining connections between woodlots and the network of interconnecting fencerows may help reduce the isolation of woodlots and fencerows.

4.4 LITERATURE CITED

Andren, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. Oikos **71**: 355-366.

Bennett, A.F. 1987. Conservation of mammals within a fragmented forest environment: the contributions of insular biogeography and autecology. Pages 41-52 *in* D.A. Saunders, G.W. Arnold, A.A. Burbidge, and A.J.M. Hopkins, editors. Nature conservation: the role of remnants of native vegetation. Surrey Beatty and Sons Pry Limited in association with CSIRO and CALM, Sydney.

Forman, R.T.T., and J. Baudry. 1984. Hedgerows and hedgerow networks in landscape ecology. Environmental Management **8(6)**: 495-510.

Gates, J.E., and L.W. Gysel. 1978. Avian nest dispersion and fledgling success in field-forest ecotones. Ecology **59(5)**: 871-883.

Haas, C.A. 1995. Dispersal and use of corridors by birds in wooded patches on an agricultural landscape. Conservation Biology **9**: 845-854.

Henderson, M.T., G. Merriam, and J. Wegner. 1985. Patchy environments and species survival: chipmunks in an agricultural mosaic. Biological Conservation **31**: 95-105.

Noss, R.F. 1983. A regional landscape approach to maintain diversity. BioScience **33(11)**: 700-706.

Stone, D.N.G. 1969. The process of rural settlement in the Athabasca area, Alberta. MA. Thesis, Department of Geography, University of Alberta.

Yahner, R.H. 1982. Avian nest densities and nest-site selection in farmstead shelterbelts. Wilson Bulletin **94(2)**: 156-175.

Yahner, R.H. 1983. Small mammals in farmstead shelterbelts: habitat correlates of seasonal abundance and community structure. Journal of Wildlife Management **47(1)**: 74-84.

Yahner, R.H. 1992. Dynamics of a small mammal community in a fragmented forest. American Midland Naturalist **127**: 381-391.

Appendix 1: Avian species recorded in fencerows during point counts in 1993 and spot mapping in 1994.

Species	*Scientific name*	*Status*[1]	*Abundance*[2] *1993*	*1994*	*Migratory status*[3]	*Habitat use*[4]	*Nesting guild*[5]	*Foraging guild*[6]
Red-tailed Hawk	*Buteo jamaicensis*	C			SD	E	ST	HP
American Kestrel	*Falco sparverius*	C			SD	E	C	HO
Ruffed Grouse	*Bonasa umbellus*	C			RES	I/E	G	G
Gray Partridge	*Perdix perdix*	C			RES	E	G	G
Northern Flicker*	*Colaptes auratus*	T		10	SD	I/E	C	G
Hairy Woodpecker*	*Picoides villosus*	C			RES	I/E	C	B
Downy Woodpecker	*Picoides pubescens*	C			RES	I/E	C	B
Eastern Kingbird	*Tyrannus tyrannus*	T		10	NTM	E	ST	A
Alder Flycatcher	*Empidonax alnorum*	C	10		NTM	E	ST	A
Least Flycatcher	*Empidonax minimus*	C	9		NTM	I/E	ST	A
Barn Swallow	*Hirundo rustica*	C			NTM	O	BG	A
Blue Jay	*Cyanocitta cristata*	C			RES	I/E	ST	G
Black-billed Magpie*	*Pica pica*	T	10	8	RES	E/0	ST	G
Common Raven	*Corvus corax*	C			RES	I	ST	G
American Crow*	*Corvus brachyrhynchos*	T	6	5	RES	E/O	ST	G
Red-eyed Vireo	*Vireo olivaceus*	T	5	3	NTM	I/E	ST	A

Appendix 1- continued

Species	*Scientific name*	*Status*[1]	*Abundance*[2] *1993*	*1994*	*Migratory status*[3]	*Habitat use*[4]	*Nesting guild*[5]	*Foraging guild*[6]
Black-capped Chickadee*	*Parus atricapillus*	T	6	3	RES	I/E	C	FG
House Wren*	*Troglodytes aedon*	T	5	7	SD	E	C	G
American Robin*	*Turdus migratorius*	T	4	6	SD	E/O	ST	G
Cedar Waxwing	*Bombycilla cedrorum*	C			SD	E	ST	F
European Starling*	*Sturnus vulgaris*	T		10	RES	E	C	G
Black-and-White Warbler	*Mniotilta varia*	C			NTM	I/E	G	B
Yellow Warbler	*Dendroica petechia*	T		10	NTM	E	ST	FG
Ovenbird	*Seiurus aurocapillus*	C			NTM	I	G	G
Common Yellowthroat	*Geothltpis trichas*	C	11		NTM	I/E	ST	FG
Mourning Warbler	*Oporornis philadelphia*	T	10	9	NTM	E	G	FG
American Redstart	*Setophaga ruticilla*	C			NTM	I/E	SR	A
Red-winged Blackbird	*Agelaius phoeniceus*	C			NTM	E	R	G
Brown-headed Cowbird	*Molothrus ater*	C			SD	E	ST	G
Northern Oriole	*Icterus galbula*	C			NTM	E	ST	FG
American Goldfinch	*Carduelis tristis*	C	12		SD	O	ST	FG
Savannah Sparrow	*Passerculus sandwichensis*	T	7	4	SD	O	G	G

Appendix 1- continued

Species	*Scientific name*	*Status*[1]	*Abundance*[2] *1993*	*1994*	*Migratory status*[3]	*Habitat use*[4]	*Nesting guild*[5]	*Foraging guild*[6]
Vesper Sparrow	*Pooecetes gramineus*	T		7	NTM	O	G	G
Dark-eyed Junco	*Junco hyemalis*	C	11		SD	E	G	G
White-throated Sparrow*	*Zonotrichia albicollis*	T	3	8	SD	E	G	G
Song Sparrow*	*Melospiza melodia*	T	1	1	SD	E	G	G
Lincoln's Sparrow	*Melospiza lincolnii*	C			NTM	?	G	G
Clay-coloured Sparrow*	*Spizella pallida*	T	2	2	SD	O	ST	G
Chipping Sparrow	*Spizella passerina*	C	8	11	NTM	E	ST	G

* Evidence for these species breeding in fencerows, nests located and fledglings observed.

[1] Status is represented by a T if at least one territory was established by that species in a fencerow in 1994 and by a C for a casual detection only.

[2] Abundances are represented by rank based on the total number of singing males detected in 1993 and the total number of territories in 1994.

[3] Species classified according to migratory status: RES, resident; SD, short distance migrant; NTM, neotropical migrant (after Ehrlich *et al.* 1988, Freemark and Collins 1989, Anon. 1991, Schmiegelow *et al.* in review).

[4] Species classified according to habitat use: E, edge; I/E, interior and edge (after Freemark & Collins 1989, Hannon pers. comm.); O, open area and/or agricultural fields (from Ehrlich *et al.* 1988), these species used fencerows predominantly for song posts with some nests found in the grass section between the fencerow and crop.

[5] Species classified according to nest location: G, ground nester; ST, shrub or tree nester; C, cavity nester; BG builbing nester; R, reed nester (from Ehrlich *et al.* 1988).

[6] Species classified according to primary foraging technique: FG, foliage gleaner; G, ground forager; A, aerial or hover gleaner; B, bark gleaner; HO, hover and pounce (from Ehrlich *et al.* 1988).

LITERATURE CITED

Anonymous, 1991. Preliminary lists of migrants for Partners in Flight Neotropical migratory bird conservation program. Partners in Flight 2: 30.

Ehrlich, P.R., D.S. Dobkin, and D. Wheye. 1988. The birder's handbook: a field guide to the natural history of North American birds. Simon and Schuster Inc., New York, USA.

Freemark, K.E., and B. Collins. 1989. Landscape ecology of breeding birds of birds breeding in temperate forest fragments. Pages 443-454 *in* J.M. Hagan III and D.W. Johnston, editors. Ecology and conservation of neotropical migrant landbirds. Smithsonian Institution Press, Washington, D.C., USA.

Schmiegelow, F.K.A., C.S. Machtans, and S.J. Hannon. In review. Are boreal birds resilient to fragmentation? An experimental study of short-term community responses. Ecology.

Appendix 2: Avian species recorded in woodlots during point counts in 1991 and 1992 (Hannon unpubl.)

Species	*Scientific name*	*Migratory status*[1]	*Habitat use*[2]	*Nesting guild*[3]	*Foraging guild*[4]
Northern Goshawk	*Accipiter gentilis*	SD	I/E	ST	AP
Red-tailed Hawk	*Buteo jamaicensis*	SD	E	ST	HP
Ruffed Grouse	*Bonasa umbellus*	RES	I/E	G	G
Mourning Dove	*Zenaida macroura*	SD	E	ST	G
Great Horned Owl	*Bubo virginianus*	RES	I/E	ST	S
Northern Flicker	*Colaptes auratus*	SD	I/E	C	G
Yellow-bellied Sapsucker	*Sphyrapicus varius*	SD	I/E	C	B
Hairy Woodpecker	*Picoides villosus*	RES	I/E	C	B
Downy Woodpecker	*Picoides pubescens*	RES	I/E	C	B
Eastern Phoebe	*Sayornis phoebe*	SD	E	BR	H
Alder Flycatcher	*Empidonax alnorum*	NTM	E	ST	A
Least Flycatcher	*Empidonax minimus*	NTM	I/E	ST	A
Western Wood-pewee	*Contopus sordidulus*	NTM	I/E	ST	A
Olive-sided Flycatcher	*Contopus borealis*	NTM	I	ST	H

Appendix 2 - continued

Species	*Scientific name*	*Migratory status*[1]	*Habitat use*[2]	*Nesting guild*[3]	*Foraging guild*[4]
Tree Swallow	*Tachycineta bicolor*	SD	E/O	C	A
Blue Jay	*Cyanocitta cristata*	RES	I/E	ST	G
Black-billed Magpie	*Pica pica*	RES	E/O	ST	G
Common Raven	*Corvus corax*	RES	I	ST	G
American Crow	*Corvus brachyrhynchos*	RES	E/O	ST	G
Solitary Vireo	*Vireo solitarius*	NTM	I	ST	FG
Red-eyed Vireo	*Vireo olivaceus*	NTM	I/E	ST	A
Philadelphia Vireo	*Vireo philadelphicus*	NTM	I	ST	H
Warbling Vireo	*Vireo gilvus*	NTM	E	ST	FG
Black-capped Chickadee	*Parus atricapillus*	RES	I/E	C	FG
White-breasted Nuthatch	*Sitta carolinensis*	RES	I	C	B
House Wren	*Troglodytes aedon*	SD	E	C	G
Gray Catbird	*Dumetella carolinensis*	SD	E	C	G

Appendix 2 - continued

Species	*Scientific name*	*Migratory status*[1]	*Habitat use*[2]	*Nesting guild*[3]	*Foraging guild*[4]
American Robin	*Turdus migratorius*	SD	E/O	ST	G
Hermit Thrush	*Catharus guttatus*	SD	I	G	G
Swainson's Thrush	*Catharus ustulatus*	NTM	I	ST	FG
Cedar Waxwing	*Bombycilla cedrorum*	SD	E	ST	F
European Starling	*Sturnus vulgaris*	RES	E	C	G
Black-and-White Warbler	*Mniotilta varia*	NTM	I/E	G	B
Tennessee Warbler	*Vermivora peregrina*	NTM	I/E	G	FG
Yellow Warbler	*Dendroica petechia*	NTM	E	ST	FG
Yellow-rumped Warbler	*Dendroica coronata*	SD	I	ST	FG
Ovenbird	*Seiurus aurocapillus*	NTM	I	G	G
Common Yellowthroat	*Geothltpis trichas*	NTM	I/E	ST	FG
Mourning Warbler	*Oporornis philadelphia*	NTM	E	G	FG
Connecticut Warbler	*Oporornis agilis*	NTM	I/E	G	G
American Redstart	*Setophaga ruticilla*	NTM	I/E	ST	A
Brewer's Blackbird	*Euphagus cyanocephalus*	NTM	E/O	ST	G

Appendix 2 - continued

Species	*Scientific name*	*Migratory status*[1]	*Habitat use*[2]	*Nesting guild*[3]	*Foraging guild*[4]
Red-winged Blackbird	*Agelaius phoeniceus*	NTM	E	R	G
Brown-headed Cowbird	*Molothrus ater*	SD	E	ST	G
Northern Oriole	*Icterus galbula*	NTM	I/E	ST	FG
Western Tanager	*Piranga ludoviciana*	NTM	I	ST	FG
Rose-breasted Grosbeak	*Pheucticus ludovicianus*	NTM	I/E	ST	FG
Purple Finch	*Carpodacus purpureus*	SD	I/E	ST	G
Pine Siskin	*Carduelis pinus*	SD	E	ST	FG
American Goldfinch	*Carduelis tristis*	SD	O	ST	FG
Savannah Sparrow	*Passerculus sandwichensis*	SD	O	G	G
Sharp-tailed Sparrow	*Ammodramus caudacutusi*	SD	O	G	G
Vesper Sparrow	*Pooecetes gramineus*	NTM	O	G	G
Dark-eyed Junco	*Junco hyemalis*	SD	E	G	G
White-throated Sparrow	*Zonotrichia albicollis*	SD	E	G	G
Song Sparrow	*Melospiza melodia*	SD	E	G	G
Lincoln's Sparrow	*Melospiza lincolnii*	NTM	E	G	G

Appendix 2 - continued

Species	*Scientific name*	*Migratory status*[1]	*Habitat use*[2]	*Nesting guild*[3]	*Foraging guild*[4]
Clay-coloured Sparrow	*Spizella pallida*	SD	O	ST	G
Chipping Sparrow	*Spizella passerina*	NTM	E	ST	G

[1] Species classified according to migratory status: RES, resident; SD, short distance migrant; NTM, neotropical migrant (after Ehrlich *et al.* 1988, Freemark and Collins 1989, Anon. 1991, Schmiegelow *et al.* in review).

[2] Species classified according to habitat use: E, edge; I/E, interior and edge (after Freemark & Collins 1989, Hannon pers. comm.).

[3] Species classified according to nest location: G, ground nester; ST, shrub or tree nester; C, cavity nester; R, reed nester (from Ehrlich *et al.* 1988).

[4] Species classified according to primary foraging technique: FG, foliage gleaner; G, ground forager; A, aerial or hover gleaner; B, bark gleaner (from Ehrlich *er al.* 1988).

LITERATURE CITED

Anonymous, 1991. Preliminary lists of migrants for Partners in Flight Neotropical migratory bird conservation program. Partners in Flight 2: 30.

Ehrlich, P.R., D.S. Dobkin, and D. Wheye. 1988. The birder's handbook: a field guide to the natural history of North American birds. Simon and Schuster Inc., New York, USA.

Freemark, K.E., and B. Collins. 1989. Landscape ecology of breeding birds of birds breeding in temperate forest fragments. Pages 443-454 *in* J.M. Hagan III and D.W. Johnston, editors. Ecology and conservation of neotropical migrant landbirds. Smithsonian Institution Press, Washington, D.C., USA.

Schmiegelow, F.K.A., C.S. Machtans, and S.J. Hannon. In review. Are boreal birds resilient to fragmentation? An experimental study of short-term community responses. Ecology.